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Limnological and Socioeconomic Evaluation of Lake Restoration Projects

Approaches and Preliminary Results

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LIMNOLOGICAL AND SOCIOECONOMIC EVALUATION OF LAKE RESTORATION PROJECTS: APPROACHES AND PRELIMINARY RESULTS

Workshop held 28 February - 2 March 1978

Project Officer

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FOREWORD

The restoration of lakes is a problem which is shared by most of the states in our nation. Numerous techniques are being used to anticipate and alleviate the impacts on water quality and the environment of competing demands on the fragile resources. The Corvallis Environmental Research Laboratory (CERL) has the responsibility of conducting environmental assessments (including limnological, social, and economic aspects) associated with lake restoration projects funded by EPA. The goal of the assessment work is to determine the effectiveness of techniques conducted on specific lakes and to compare the different techniques employed. In conjunction with these goals, CERL is developing improved methodologies to assess the limnological, social, and economic impacts, both positive and negative, of lake restoration.

A workshop was conducted, with the assistance of the Water Resources Research Institute, Oregon State University, to bring together all persons directly associated with the EPA Clean Lakes Program in order to describe the overall program; to discuss in depth the evaluation techniques being used; and to explore various decision criteria concerning lake restoration projects. The papers presented during the three-day workshop are published in this volume.

James C. McCarty
Acting Director
CERL

ABSTRACT

A total of 19 papers was presented at the workshop held 28 February - 2 March, 1978 on the campus of Oregon State University. The objective was to assemble grantees and project officers associated with EPA's Lake Restoration Evaluation Program so that they could become familiar with each other's work. Outside experts were invited to offer constructive criticism of the current approach to assessment of techniques. Several lakes were considered for limnological, social and economic aspects. A draft copy of the Lake Evaluation Index (LEI) developed by EPA was presented and discussed.

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This conference was hosted by the Environmental Protection Agency and Oregon State University. Arrangements on campus were made by Peter C. Klingeman and William H. Buckley both of the OSU Water Resources Research Institute. They were instrumental in assembling and compiling the papers which contributed to this proceedings. Their assistance is appreciated greatly. Thanks goes also to the participants for sharing their planned approaches, some of their preliminary data, and for their willingness to commit some of these preliminary ideas to writing so that each of them might reevaluate his own approach based on the ideas of others in the program. Appreciation is extended to our invited experts Donald Porcella, Charles P. Wolf, Herbert Stoevener, Greg J. Protasel and Thomas Crocker who listened attentively, then offered suggestions for improving various aspects of our program. Special thanks goes to Dr. Wolf who supplied lengthy written comments concerning our attempts to develop a truly integrated social, economic and limnological lake restoration assessment program.

OVERVIEW OF EPA LAKE RESTORATION EVALUATION PROGRAM

by

S. A. Peterson

The Clean Lakes Program was originally initiated by the Office of Water Planning and Standards (OWPS) as a demonstration grant program. Demonstration grants are similar to research grants but they essentially involve an experiment under actual conditions. The choice to implement the program in this manner was understandable. Most of us in this room would probably agree that lake restoration is an inexact though developing science. The truth of the matter is that we can predict with relative certainty the general direction of change in a lake resulting from most known types of lake restoration techniques. What we have been unable to do with any precision is to predict quantitatively the degree of change in general usability of a lake affected by inlake or watershed treatment techniques, either individually or in combination with each other. Currently, there is no reliable method available for determining the optimum treatment technique for specific lakes or groups of similar lakes. Furthermore, the impact of lake restoration on the lake community *per se* and the surrounding complex social structure is nearly impossible to predict.

These were some of the reasons why, in 1975, the OWPS requested the assistance of the Office of Research and Development (ORD) to assess and evaluate the effectiveness of lake restoration demonstrations being conducted under the auspices of Section 314 of Public Law 92-500. EPA's Corvallis Environmental Research Laboratory (CERL) was assigned the responsibility for planning and conducting the evaluation program. A major constraint of the evaluation program was that projects had to be selected from previously funded 314 demonstration projects. I will point out later why this presented a problem.

At the outset CERL envisioned two major objectives for the Clean Lakes Evaluation Program. These were 1) to determine the effectiveness of specific restoration techniques or combination of techniques on specific lakes and 2) to compare the effectiveness of various restoration processes on different lakes. These evaluations were to include not only the commonly measured limnological variables but also various aspects of the economic and sociological impacts associated with lake restoration. Funds to accomplish these goals (to date $2.1M) have come directly from the 314 program and were set aside by the OWPS specifically for the evaluation projects. None of these funds were from the ORD, the usual funding mechanism for the research laboratories. All clean lakes evaluation funds were designated for extramural expenditure which meant the evaluation program would be conducted through grants or contracts. Because of their greater flexibility and the shorter time frame required to negotiate grants it was decided to use that mechanism rather than contracts to fund evaluations on specific lake resotration projects.

These grants basically will satisfy the requirements for objective number one, "Determination of the effectiveness of specific restoration manipulations on specific lakes." Grants on specific projects in themselves, however, will not satisfy objective number two, "Comparison of effectiveness of various restoration techniques on different lakes." Therefore, CERL is paralleling the extramural effort, and hopes to use data from that effort, with a modest, inhouse, ORD-funded clean lakes evaluation effort which has two objectives of its own. They are 1) to develop methods which will improve our capabilities to predict the response of lakes to restorative manipulations and 2) to develop a lake restoration guidance or user's manual to assist Federal, State and local water resource managers with decisions concerning lake restoration and techniques for assessing the environmental effects, both socioeconomically and limnologically, of lake restoration.

Funding limitations made it impractical to attempt to evaluate each lake restoration project in detail. Therefore, the strategy was to devise a means for evaluating a subset of lakes which was representative of the entire set in terms of treatment technique, watershed types, geographic distribution and sociological setting. All of the nearly 60 projects funded as of September, 1976, were categorized according to their primary mode of treatment. The categories included source control, in-lake control and problem treatment techniques (Table 1). When classified by this method it quickly became evident that a problem existed from the standpoint of evaluating individual lake restoration techniques. Only 12 of the restoration projects had single manipulations. On the average each project had 2.3 restoration techniques being applied. Ideally one would like to look at single manipulations in order to evaluate them; however, since we had to select from previously funded projects that was not our prerogative. Therefore, inasmuch as one of our objectives was to evaluate the effectiveness of combinations of restoration techniques anyway, we proceeded to select carefully a subset of projects that was representative of the set as a whole.

By classifying all of the restoration projects according to one of three major "lake restoration techniques" it was possible to group the many manipulations into a limited number of similar types and to approach the set of lakes in the manner of an experimental design. The experimental design depends on three assumptions. These are that 1) treating manipulations in terms of their effect is a valid approach, 2) that different lakes can be "standardized" through the use of mass-balance models similar to Vollenwider or Dillion and Rigler and 3) that the relative quantitative impacts of the manipulations can be determined.

All of the 314 projects were ranked according to 1) the quality of the baseline data available, 2) the length of time and frequency of baseline data collection, 3), the potential for quantification of changes in phosphorus loading on a short and long term basis, 4) probability of measurable short-term response of the lake and 5) the number of manipulations. Assuming that the overall approach of the experimental design was valid for the set of lakes and manipulations that had been funded, the top ranking projects were treated, according to manipulation type, as a standard factorial design. Eighteen projects were plugged into the design which resulted in the configuration shown in Table 2. The Clean Lakes work up to this point resulted in publication of CERL report number 034 (Porcella and Peterson, 1977). Delays in implementation of some restoration grants or too rapid implementation (before adequate baseline data could be compiled) have resulted in some deletions and/or substitutions of projects originally identified in the design matrix (Table 3). Lake restoration projects currently being evaluated are shown in Table 4.

These lakes and their role in the surrounding social setting form a complex limnological-social system in which it is impossible to predict what the effects of any particular restoration effort will be, on either the lake or the community of current and potential users. Thus, the Clean Lakes Evaluation Program is viewed as a means of both identifying the most useful and cost effective lake restoration techniques for future projects and as an opportunity to enhance that information by obtaining a better understanding of the limnological and social impacts resulting from such restorations (Christiansen, 1978).

The basic concept of the Clean Lakes Evaluation Program is still to assess the effectiveness of individual and combinations of different restoration techniques on specific lakes and to compare their relative efficiencies to one another. However, to know what impact the restoration treatment has had on the lake itself is not enough. The idea of lake restoration, after all, is a human concept, with its costs and benefits supposedly weighed against those of other projects in the community, or watershed, which available funds might be used for. Therefore, it is extremely important to be able to transform the results of lake restoration, in terms of limnological alterations, into meaningful and useful social and economic pieces of information. In the final analysis, the success or failure of a lake restoration in general, will be determined on the basis of how it is perceived by the public which uses and pays for the improved facilities.

A variety of lake restoration treatment techniques are being employed in the Clean Lakes Program with a number of different objectives. The one thread of commonality woven among the projects we have selected, however, is that all are directed toward reducing the phosphorus supply to the lakes. Therefore, one approach being used to assess the effects of the various treatments is chlorophyll a-phosphorus mass balance modeling. You have had an opportunity to examine Phil Larsen's paper on this subject and he will be presenting it to the group tomorrow.

Another way to assess the response of lakes to a restoration manipulation is to measure changes in a number of key variables and combine them in a way that permits a comparison of the relative effectiveness of the different treatment techniques. This line of thought resulted in the draft Lake Evaluation Index (LEI) developed by Don Porcella and me (Porcella and Peterson, 1977). Ron Glessner, however, has been working with the LEI since last fall so he will make a presentation tomorrow on the LEI itself and some of the work he has done in an attempt to test its validity.

The socioeconomic aspects of the evaluations are dependent on limnological quantification of changes in the lake system due to the restoration treatment. Limnological information will be employed to develop user demand functions for the purpose of predicting the number and type of users as a function of the restoration treatment. The socioeconomic gains and losses to individuals and the lake community as a whole will be measured. Also the direct and maintainance costs of various restoration treatments will be determined.

All of the in-house approaches to lake restoration assessment are highly dependent on data from, and the full cooperation of, our grantees. Mass balance modeling, the LEI, and the development of user demand functions are not ends in themselves. They are means to an end. The final objective of our assessment program is to put into the hands of decision makers a lake restoration handbook or guide which will assist them in predicting the consequences of employing various restoration techniques. It will be compiled through our interpretation of models, the LEI and the user demand functions. With this information in hand a decision can be made as to whether or not to go ahead with a restoration project, as well as how to proceed and how to assess the outcome if the project is begun.

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Christiansen, Neils. Social Evaluation of the Clean Lakes Progam: A Research Strategy. Mimeo. 21 pp. 1978.

Larsen, David P. Evaluation of Clean Lakes Restoration Using Phosphorus Mass Balance Modeling. Draft Report. Feb. 1978.

TABLE 1. CLASSIFICATION OF LAKE RESTORATION TECHNIQUES

I. Source Controls

  A. Treatment of inflows

  B. Diversion of inflows

  C. Watershed management (land uses, practices, nonpoint source control, regulations and/or treatments).

  D. Lake riparian regulation or modification

  E. Product modification or regulation

II. In-Lake Controls

  A. Dredging

  B. Volume changes other than by dredging or compaction of sediments.

  C. Nutrient inactivation

  D. Dilution/Flushing

  E. Flow adjustment

  F. Sediment exposure and dessication

  G. Lake bottom sealing

  H. In-lake sediment leaching

  I. Shoreline modification

  J. Riparian treatment of lake water

  K. Selective discharge

III. Problem Treatment (directed at biological consequences of lake condition)

  A. Physical techniques (harvesting, water level fluctuations, habitat manipulations)

  B. Chemical (algicides, herbicides, piscicides)

  C. Biological (predator-prey manipulations, pathological reactions)

  D. Mixing (aeration, mechanical pumps, lake bottom modification)

  E. Aeration (add DO; e.g. hypolimnetic aeration)

TABLE 2. EXPERIMENTAL DESIGN OF CANDIDATE LAKE RESTORATION EVALUATION PROJECTS

	SOURCE	INLAKE			OTHER
		DREDGING	NUT. INACT	DIL/FLUSH	
SOURCE	Fountain, MN RonKonKoma NY Clear, MN White Clay, WI Loch Raven, MD	Long, MN	Mirror/ Shadow, WI Liberty, WA		
DREDGING		Lansing, MI Muskego, WI Collins Park, NY Lenox, IA	Lilly, WI	Vancouver, WA	
INLAKE NUT. INACT			Medical, WA Lafayette, CA		
DIL/ FLUSH				Moses, WA	
OTHER					Long(Kitsap CO), WA (drawdown, dredge, nut. inact, NPS)

TABLE 3. PROJECTS DELETED AND/OR SUBSTITUTED FROM THE ORIGINAL SET OF CANDIDATE LAKE RESTORATION EVALUATION PROJECTS

		Substituted	
Project Type	Deleted	Original	Current
Source Control	Fountain Lake, MN	Clear Lake, MN	Long Lake, MN
Inlake Control	Lenox Lake, IA Medical Lake, WA*		

* Implementation grant funds used to assess the effects of the restoration project.

TABLE 4. CURRENTLY ACTIVE EVALUATION PROJECTS

Lake	Evaluation Type		Restoration Control Type			Principal Investigator	
	Limno	Socio/Econ	Source	Inlake	Other	Limno	Socio/Econ
Mirror/Shadow, WI	X	X	X	X		Douglas Knauer	Lowell Klessig
Liberty Lake, WA	X	X	√	X		Bill Funk	Tom Hogg Bill Honey Ken Gibbs
RonKonKoma, NY	√	X	X			*	Ben Lieu
White Clay, WI	X	X	X			Jim Peterson	Lowell Klessig
Long Lake, MN	X	X	X	√		Roger Blomquist	Ben Lieu
Lansing, MI	X			X		Cal McNabb	
Skinner Lake, IN	X		X			Cal McNabb	
Collins Park, NY	X			X		Carl George	
Lilly Lake, WI	X			X		Russel Dunst	
Lafayette, CA	X			X		Marc Lorenzen	
Moses Lake, WA	X			X		Gene Welch	
Long Lake, WA	X			X	X	Gene Welch	

X = A thorough evaluation of a major manipulation.
√ = A less comprehensive evaluation or a less significant manipulation.
* = Limnological evaluation is part of the demonstration grant.

A STATUS REPORT ON THE MIRROR/SHADOW LAKES EVALUATION PROJECT

by

D. R. Knauer and P. J. Garrison*

The objective of any lake renewal project is to restore lakes, which have experienced an increased rate of eutrophication owing to cultural processes in the watershed, to a less deteriorated and more useful state. It is incumbent on those proposing lake protection and restoration alternatives to have developed first a comprehensive hydraulic and nutrient budget for a particular lake. Such considerations determine to a large extent the feasibility of treatment and control strategies necessary for a given set of problems.

In determining the sources of lake degradation for Mirror and Shadow Lakes, complete hydraulic and nutrient budgets were completed to include the contributions of groundwater, precipitation, diffuse runoff, and storm water influx to these lakes. The nutrient loadings from the various compartments have identified the urban storm water runoff as the major contributing source of phosphorus to these lakes.

BACKGROUND DATA

Drainage Basin Characteristics

The Mirror and Shadow Lake basins are within the City of Waupaca.

The lake basins are "kettle holes" in outwash plains formed during the recession of the Cary ice sheet during Pleistocene glaciation. In the vicinity of Mirror Lake, the outwash consists of a 15 to 30 m thick sequence of medium to coarse-grained sand with gravel lenses and overlies 15 m of glacial till, which in turn rests on granite bedrock.

The Waupaca area was settled in the 1850's. The population of 2500 in 1885 increased to about 4400 in 1970. Streets around Mirror Lake were built by 1901, and residential building in the vicinity was nearly complete by

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* Office of Inland Lake Renewal, Wisconsin Department of Natural Resources, Madison, Wisconsin 53707.

1934. Storm sewerage (and street paving) was installed in the 1920's. The city pumped drinking water directly from Mirror Lake from 1908 until 1913 at which time a shallow well was constructed near the lake.

Figure 1 shows the storm water drainage system and entry points of effluent into the lakes before diversion. The storm drainage basins for Mirror Lake total about 60 percent of the estimated natural surface water drainage, the residual area is the land extending from the perimeter of the lake to the nearest street. The basin immediately north of Mirror Lake covers approximately 17 hectares primarily of established residential dwellings. The storm sewer received no continual water input and thus provided characteristics of urban runoff with little interference from ground waters. A bubble gage stage recorder system was installed and maintained by the U.S. Geological Survey to determine flows in the 53 cm diameter concrete pipe sewer leading to the north shore.

The area of the basin east of Mirror Lake was 2 hectares in size. The limits of the drainage area near the two western-most inlets are poorly defined, being surrounded by grassed areas and not directly connected to the gutter system on the nearby street. The U.S. Geological Survey built and maintained a waterflow gaging station near the lake employing an 18-inch, 90° V-notch weir.

The monitored drainage basin for Shadow Lake incorporated about 20 hectares of developed urban land to the north and east and about 36 hectares of undeveloped lowlands surrounding an intermittent (former trout) stream. The stream serves as an open channel conduit for storm water flow. The base flow of the stream is about 0.02 cfs. The drainage basin incorporated about 75 percent of the estimated surface water drainage area for Shadow Lake. Stream flows were estimated from stage recordings at a U.S.G.S. H-flume installation.

Sample collection from the North Mirror Basin was facilitated by an automatic sequence sampler which could be programmed to take samples at intervals as short as 10 minutes. Vacuum operated samplers paced at 30 minutes were used at the other two gaging installations.

Figure 2 shows precipitation, accumulative storm water flow and snow cover in the Mirror Lake basin for 1972.

Description of Lakes

The physical description of Mirror Lake (Figure 3) is as follows:

Area	5.1 ha
Volume	4 X $10^5 m^3$
Mean Depth (V/A)	7.8 m
Maximum Depth.	13.1 m

and *Shadow Lake* (Figure 4):

Area	17.1 ha
Volume	9.1 x $10^5 m^3$
Mean Depth (V/A)	5.3 m
Maximum Depth	12.4 m

Based on the hydrologic budgets and lake volumes, the theoretical hydraulic residence time for Mirror Lake was 4.3 years and for Shadow Lake, 2.1 years.

The annual transport of material to Mirror and Shadow Lakes via the storm sewers is presented in Tables 1 and 2. The amount of materials transported to Mirror Lake is considered more detrimental to that ecosystem, owing to the smaller surface area and volume. The various sources of phosphorus loading to Mirror Lake for 1972 and 1973 are presented in Table 3. Applying Vollenweider's criteria for permissible and dangerous phosphorus loadings to the storm sewer influx only, it is apparent that Mirror Lake was seriously being stressed. Figures 5, 6 and 7 illustrate the phytoplankton response to large amounts of nutrient influx via the storm sewers during an unusually "wet" August and September, 1972. Primary productivity increased from 395 $mgC/m^2/day$ to 1415 $mgC/m^2/day$ and the phytoplankton biomass responded in a similar manner by increasing 5.5 mm^3/l. The temporal succession in major phytoplankton taxa changed from dominance by *Cyclotella*, *Chroococcus* and genera of Chlorophyta before the nutrient influx in August and September to dominance by *Anabaena* and *Oscillatoria* following the nutrient enrichment of the euphotic zone.

Figures 8 and 9 illustrate the present distribution of aquatic macrophytes in Mirror and Shadow Lakes. Tables 4 and 5 represent the frequency occurrence of macrophytes for Mirror and Shadow Lakes.

Inlake sedimentation rates have been calculated for Mirror Lake using radiometric dating of a sediment core taken from the middle of the lake. Using ^{210}Pb, the rates of sedimentation over the past 100 years are approximately 0.064 cm/yr and approximately 0.26 cm/yr recently (Figure 10). These data suggest an increase in the rate of sedimentation occurred about 1945. This increase in sedimentation may be owing to increased construction activities in the watershed as a part of the post World War II building boom. This rationale is, of course, conjecture on our part.

Oscillatoria rubescens, a blue-green alga, has been observed in Mirror Lake since at least 1950. Until 1950, Mirror Lake was a source of block ice for commercial use, however, the operation was discontinued owing to discoloring and a smell that was associated with the melting ice. In 1971-73, we observed *O. rubescens* was in significant amounts in the surface waters during late fall and early winter to cause problems. In 1976 we took a sediment core from the middle of the lake for the purpose of analyzing for oscillaxanthin. Figure 11 shows the oscillaxanthin pigment with depth of core, and it is apparent that *O. rubescens* has been present in the lake in significant biomass since 1950.

EVALUATION APPROACH

SPECIFIC EVALUATION OBJECTIVES

The specific evaluation objectives for this program are:

1. Nutrient Diversion - The objective is to monitor physiochemical and biological responses to nutrient diversion. Data collected during 1972 and 1973 indicated the possible consequence of storm sewer diversion would be lower phytoplankton biomass and productivities. Actual data will be applied to existing modelling efforts in an attempt to verify predictive models.

2. Nutrient Inactivation/Precipitation - Alum will be applied to the hypolimnion during the early summer of 1978. The hypolimnetic application of alum to the lakes may not affect the conditions in the euphotic zone until fall overturn of 1978, therefore, the evaluation of the ecological processes which occur in the euphotic zone should not be altered drastically. Examination of past data indicates that metalimnetic oscillations are not occurring at a significant amplitude to supply inorganic -P into the euphotic zone. The alum treatment will be evaluated in several ways to answer questions of the effectiveness of the Al-floc in retarding sediment P release.

   (a) Sediment P release will be measured "in-situ" under normal hypolimnetic conditions and also under abiotic conditions before and following the alum addition. "In-situ" rates will be compared.

   (b) Sediment cores will be taken before the treatment and Al concentrations measured. During the 2 years following treatment, sediment cores will be taken to examine the distribution pattern of the aluminum with time.

   (c) Seston traps will be placed in the hypolimnion to collect sedimenting organic material to evaluate the potential P available for release as the alum floc becomes overburdened with falling organic matter. Our data from other alum treatments suggest this may be a possible mechanism for resolubilization of P in the hypolimnion of alum treated lakes.

PROGRAM STATUS

METHODS

Water samples for physico-chemical and biological analyses were collected from the deepest portion of both lakes at monthly intervals from October 1976 until March 1977; and from March through October 1977 water samples were collected at biweekly intervals.

Water chemical analyses were conducted at the State Laboratory of Hygiene usually within 24 hours of collection. Analytical methods used were those in the EPA 1974 publication Methods for Chemical Analysis of Water and Wastes. The phosphorus and nitrogen chemical species were analyzed by auto-analyzer procedures.

Chlorophyll a was determined using the 90 percent acetone extraction method. Samples were filtered immediately upon collection through a glass fiber filter and allowed to extract for a minimum of 24 hours at a temperature of less than 0°C. The filters were then ground to a fine pulp and allowed to extract at least overnight. Absorption was measured with a Bausch and Lomb Spectronic 70 with a slit width of 8mm. Since this machine underestimates the chlorophyll a absorption peak, a correction factor of 1.11 was applied. Chlorophyll a was computed by the phaeophytin correction method of Strickland and Parsons (1972).

Primary production was measured by the ^{14}C method, starting on May 11, 1977, in Mirror Lake. Samples were collected at one meter intervals between 0.5 m and 7.5 m. Samples were inoculated with approximately 6 μCi of $NaH^{14}CO_3$, and incubated from 10:00 a.m. to 2:00 p.m. CST at the depths taken. The bottles were suspended horizontally under a bar between two floats in order to minimize artificial shading. After incubation the samples were immediately placed in the dark and either placed on ice or treated with 1 ml of 1 percent merthiolate. In order to determine the amount of ^{14}C fixed, an acid-bubbling technique modified from Schindler et al (1972) was used until July 18. When this method proved unsatisfactory, the samples (usually 100 ml) were filtered through a 47 mm membrane filter (0.45 μpore size). The filters were washed with a minimum of 200 ml of distilled water and immediately dissolved in a liquid scintillation fluor described by Schindler (1966). The radioactivity was measured with a liquid scintillation spectrophotometer and the results incorporated into the equation in Standard Methods (APHA, 1971). The initial inorganic carbon was determined from total alkalinity, pH, and temperature using the equations of Rainwater and Thatcher (1959). Daily photosynthesis was computed by the method of Schindler and Nighswander (1970).

The phytoplankton biomass was determined by approximating the geometric forms of individual phytoplankters from samples preserved with acidic Lugol's solution, and applying this volume to total cell counts made by a modified Utermohl method as described by Lund et al. (1958). Samples were collected from either 2 or 2.5 m.

Water transparency readings were taken with a 20 cm white Secchi disc. Submarine light measurements were made with a G.M. submersible light meter on several occasions. Incident solar radiation was measured with a model R411 Star Pyranometer (Weather Measure Corp.) connected to a potentiometric strip chart recorder. These instruments were located adjacent to Mirror Lake.

Nutrient regeneration chambers similar to those used in the Shagawa Lake study were placed on the sediments of Mirror Lake in June, 1977 (See Sonzogni et al. for description). Water samples were taken from within the chambers near the sediment surface and near the top by scuba divers using a 500 ml stainless steel syringe.

RESULTS AND DISCUSSION

Several studies have described the high nutrient content of storm water runoff and its relationship to nutrient loading (McGriff, 1972; Kluesener and Lee, 1974; Vitale and Sprey, 1974; and Knauer, 1975). Although other nutrients are in storm water runoff, phosphorus has generally been found to be the limiting factor in many lakes, and consequently, phosphorus loading directly affects phytoplankton production (Vollenweider, 1968; Edmondson, 1972; Schindler et al, 1973; and Jones and Bachmann, 1975).

Table 6 presents annual P loading rates for 1972, 1973 and 1977. With the diversion of the storm sewers, the P loading was reduced from a value of 0.418 g P/m^2/yr to 0.213 g P/m^2/yr. The average in lake annual total phosphorus concentrations for Mirror Lake for the years 1972, 1973 and 1977 were 0.088 mg/l, 0.093 mg/l and 0.090 mg/l respectively. Although the data suggest that the phosphorus concentration has not changed since diversion, a one-year time period is not sufficient for the lake to respond to a new steady-state condition.

Total phosphorus during the ice covered period was present in concentrations of 50 µg/l in the surface waters. During the summer months, concentrations decreased to 20 µg/l in the upper euphotic zone and remained at 50 µg/l in the lower euphotic zone (Figure 12) with much higher P concentrations in the hypolimnion.

Dissolved reactive phosphorus in the surface water was present in undetectable concentrations, 4 µg/l, during much of the year (Figure 13). In the hypolimnion, concentrations were over 500 µg/l.

Since it is unclear the role sediments have on the internal phosphorus loading, nutrient regeneration chambers were placed "in-situ" on the bottom of Mirror Lake. Phosphorus and ammonium-nitrogen trends inside the chambers are illustrated in Figures 14 and 15. Total phosphorus in the chambers did not increase until 50 days after the chambers were placed on the sediments, even though the environment within the chambers was anoxic within 20 days. The time response before phosphorus release from the sediments was observed, was contrary to the results of Sonzogni et al. (1977) and observations from inlake measurements. This initial time lag was not noticed with ammonium-nitrogen, however, as increases were observed prior to the development of an anoxic environment.

The phosphorus concentrations in the chamber increased from 190 µg/l to 580 µg/l over a period of 157 days. Phosphorus release rates were calculated during anoxic conditions from July 26 through September 27. The rate of phosphorus release from sediments during this time period was 1.8 mg/m^2·day. This release rate was lower than expected for an eutrophic lake. Phosphorus release rates for eutrophic lakes reported in literature range from 4.0 mg/m^2·day for Lake Sammamish (Welch and Spyridakis, 1972) to 10.8 mg/m^2·day for Lake Mendota (Sonzogni, 1974).

Ammonium-nitrogen release rates under anoxic conditions were 11.6 mg/m^2·day. This rate is similar to anaerobic release rates observed from other eutrophic lakes, e.g., Lake Furesø, 11.1 mg/m^2·day and Lake Esrom, 13.1 mg/m^2·day (Kamp-Nielsen, 1974).

Temperature and dissolved oxygen isopleths (Figures 16 and 17) show the lake did not completely mix during the fall of 1976 and spring of 1977. A metalimnetic oxygen maxima was present throughout the summer owing to algal photosynthesis.

The depth of Secchi disc measurements is often used as an indication of algal standing crops (Dillon and Rigler, 1975 and Carlson, 1977). The validity of this relationship is dependent upon other causes of water turbidity in addition to phytoplankton. The correlation coefficient between Secchi disc depth and chlorophyll a concentration for Mirror Lake was 0.92 which is significant at the 95 percent level (Figure 18).

Secchi disc transparency measurements were generally less in the spring than the summer (Figure 19). This was due primarily to the presence of the alga Oscillatoria rubescens. As O. rubescens disappears from the surface waters, the Secchi disc depths correspondingly increased. The average summer Secchi disc depths (June-September) were similar before and after storm sewer diversion, approximately 3.4 m. The minimum Secchi disc depths (0.9 m) were also similar for 1972, 1973 and 1977, and occurred when O. rubescens was the dominant phytoplankter in the surface waters. The maximum Secchi disc depths varied from 5.2 m in 1972 to 4.3 m in 1973 to 4.7 m in 1977.

Chlorophyll a concentrations were much higher during April 1977 than at any time during the year (Figure 20). The high chlorophyll concentrations were owing primarily to the abundance of Oscillatoria rubescens. By the end of April, O. rubescens started to sediment to the deeper water, and by May was found only in 5-7 m strata. During the summer, chlorophyll a values decrease in the deeper waters from a high 80 µg/l to 33 µg/l by mid September. O. rubescens was present at a depth during the summer where the light levels were generally below 1 percent of incident light. In the surface waters, during the summer, chlorophyll a values were low, 1-4 µg/l, and the chlorophyll concentrations showed little fluctuation (Figure 20). In October chlorophyll a values were generally evenly distributed throughout the water column and coincided with the distribution of O. rubescens.

Although changes in the limnology of Mirror Lake after storm sewer diversion are not evident from the phosphorus chemistry data or water transparency, there appeared to be a change in the phytoplankton. Primary productivity depth profiles for the summers 1972 and 1977 are compared in Figure 21. Knauer (1975) indicated that the low production from May to early August 1972 was owing to reduced storm water runoff as a result of a below normal rainfall during those months and the fact that the lake failed to mix the previous spring and fall. This was also indicated by the Secchi disc depths. However, the importance of the storm sewer input to the lake is evident from August through October when, with increased rainfall (Table 7), the production curves changed drastically. Whereas the curves resembled those from oligotrophic or mesotrophic waters before August, with the increased rainfall the curves took on the appearance of those most likely found in eutrophic lakes. The production curves for 1977 remained similar to those expected in mesotrophic waters even though rainfall was greater in 1977 than in 1972 during most of the summer months. This emphasizes the effects of the lack of storm sewer discharge to provide the necessary nutrients to stimulate phytoplankton productivity. Maximum primary productivity levels in 1977 were only half of those experienced in 1972 (32.6 mg C/m^3/hr and 60.5 mg C/m^3/hr, respectively).

Primary productivity was also compared on an areal basis for Mirror Lake (Figure 22). Maximum daily rates were similar for both years, being 1.415 g C/m^2/day in 1972 and 1.297 g C/m^2/day in 1977.

Algae belonging to the division Cyanophyta dominated the phytoplankton during the spring and fall in 1977 (Figure 23). _Oscillatoria rubescens_ was the dominant alga at that time. During the summer growing season, green algae generally dominated. These were mostly colonial species such as _Oocystis pusilla_, _Sphaerocystic Schroeteri_ and _Gloeocystis plactonica_, although the bacillariophyta were an important part of the phytoplankton crop on June 20.

The species composition of the phytoplankton in 1977 was different than before the storm sewer diversion (Figure 23). Knauer (1975) reported that after nutrient enrichment from the storm sewers, the blue-green algae _Anabaena_ sp. and _Chroococcus_ sp. dominated the phytoplankton. These genera were either not present or only in very low numbers in 1977.

The phytoplankton biomass from May through September, 1977, did not exceed 2.0 mm^3/l (Figure 24). The biomass increased to 3.5 mm^3/l in October, probably in response to the deepening of the metalimnion.

SHADOW LAKE

Dissolved oxygen and temperature isopleths from 1976-77 are shown in Figures 25 and 26). In the past, limnological studies on Shadow Lake were not as intensive as for Mirror Lake. Secchi disc measurements were taken on a regular basis only during the summer of 1971. The depth of Secchi disc measurements was greater during the summer of 1977 when compared to 1971 data, 3.2 m and 1.9 m respectively (Figure 27). The correlation coefficient between chlorophyll _a_ and Secchi disc measurements was 0.78 (Figure 28).

During 1977, the chlorophyll _a_ concentrations were highest in April (Figure 29). _Oscillatoria rubescens_ was the dominant phytoplankter during this period. As _O. rubescens_ sediments to the deeper waters, chlorophyll _a_ concentrations decreased to approximately 5 μg/l in the surface waters throughout the summer. As _O. rubescens_ became distributed throughout the epilimnion during the fall, chlorophyll _a_ concentrations increased correspondingly.

Isopleths of total phosphorus concentrations are presented in Figure 30. The highest concentrations, ca. 470 μg/l, were observed in the hypoliminon during the summer. Total phosphorus concentrations in the epilimnion were generally about 20 μg/l, similar to Mirror Lake.

Table 8 lists the average concentrations for selected chemical parameters from October 17, 1976 through October 25, 1977.

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TABLE 1. CONCENTRATIONS AND TRANSPORT OF MATERIALS IN URBAN RUNOFF -- MIRROR LAKE BASINS NORTH AND EAST, 1972 (FROM PETERSON AND KNAUER, 1978)

BASIN		NORTH			EAST	
Area (ha)		17			2.1	
Precipitation (cm)		77			77	
Runoff ($m^3 \times 10^3$)		194			55	
Runoff %		14.8			35.2	
Street and Parking Lot Area %		21			25	
	Mean Concentrations[a]	Total kg	Output ($g/m^2/yr$)	Mean Concentrations[a]	Total kg	Output ($g/m^2/yr$)
Total Phosphorus	0.41	8.0	0.047	0.33	1.8	0.088
Reactive P (est.)	0.14	2.7	0.016	0.11	0.6	0.029
Total N	2.55	49.4	0.29	1.46	8.0	0.39
Inorganic N	1.30	25.2	0.15	0.57	3.1	0.16
Organic N	1.25	24.3	0.14	0.90	4.9	0.24
BOD_5	11.3	220	1.3	9.0	48.5	2.4
Cl	78	1,506	8.8	17	95.7	4.7
Na	36	698	4.2	14	76.6	3.7
K	3.6	70.3	0.42	2.6	14.4	0.71
Ca	12	231	1.4	11	59	2.9
Mg	5.9	115	0.67	6.1	34	1.7
Total Solids	325	6,305	37	240	1,343	66

[a] in mg/l except as noted.

TABLE 2. SUMMARY OF STORM SEWER MATERIAL TRANSPORT SHADOW LAKE BASIN, 1972 (FROM PETERSON & KNAUER, 1978)

Area - (ha) = 58.6; Precipitation - (cm) = 77

Runoff - $m^3 \times 10^3$ = 67, Runoff % = 14.8; Street & Parking Lot Area % = 7.6

	Mean Concentrations[a]	Total kg	Output ($g/m^2/yr$)
Total Phosphorus	0.22	14.4	0.025
Reactive P (est.)	0.07	4.8	0.0078
Total Nitrogen	2.17	146	0.25
Inorganic N	0.92	62	0.105
Organic N	1.25	84	0.14
BOD_5	5.1	340	0.58
Cl	213	14,243	24
Na	100	6,713	11
K	4.0	268	0.46
Ca	52	3,447	5.9
Mg	26	1,724	2.9
Total Solids	430	28,576	49

[a] in mg/l except as noted.

TABLE 3. TOTAL PHOSPHORUS LOADING RATES TO MIRROR LAKE, 1972 and 1973

	g/m²/yr 1972	g/m²/yr 1973[a]	Vollenweider's Loading Rates Permissible	Vollenweider's Loading Rates Dangerous
Rainfall	0.053	0.016		
Storm Sewer	0.190	0.261		
Diffuse	0.060	0.097		
Ground Water	0.080	0.080		
TOTAL	0.383	0.454	0.088	0.172
Total Without Storm Sewer	0.193	0.193		
Total Without Ground Water And Storm Sewer[b]	0.113	0.113		

[a] Estimated from precipitation, storm sewer flow data and 1972 runoff coefficients from May-November.

[b] Total without ground water as well as storm sewer is included because Vollenweider's criteria were established without regard for possible nutrient influx via ground water into the lakes.

TABLE 4. FREQUENCY OCCURRENCE OF AQUATIC MACROPHYTES IN MIRROR LAKE

Species	Percent[1] Frequency Occurrence	Relative[2] Frequency Occurrence
Myriophyllum exalbescens	84.0	19.2
Ceratophyllum demersum	79.7	18.2
Potamogeton pusillus	57.2	13.1
Potamogeton pectinatus	53.5	12.2
Chara spp.	34.8	8.0
Heteranthia dubia	32.1	7.4
Anacharis canadensis	29.9	6.9
Potamogeton zosterformis	24.6	5.6
Potamogeton alpinus	15.5	3.6
Vallisneria americana	12.8	2.9
Nymphaea tuberosa	8.6	2.0
Najas flexilis	3.7	0.9
Numphar variegatum[3]		
Potamogeton natans[3]		
Any species above	100.0	100.0

[1] No. of occurrences in BSU/total BSU.
[2] No. of occurrences in BSU/total species encountered in all BSU's.
[3] Present but not found in a BSU.

TABLE 5. FREQUENCY OCCURRENCE OF AQUATIC MACROPHYTES IN SHADOW LAKE.

Species	Percent Frequency Occurrence	Relative Frequency Occurrence
Chara spp.	74.3	19.3
Potamogeton alpinus	66.3	17.2
Myriophyllum exalbescens	46.1	12.0
Potamogeton pectinatus	44.0	11.4
Potamogeton pusillus	24.3	6.3
Ceratophyllum demersum	23.0	6.0
Heteranthia dubia	19.9	5.2
Vallisneria americana	19.2	5.0
Najas flexilis	16.6	4.6
Potamogeton zosteriformis	13.2	3.4
Nymphaea tuberosa	12.4	3.2
Anacharis canadensis	11.1	2.9
Potamogeton nodosus	9.3	2.4
Potamogeton natans	3.9	1.0
Numphar variegatum	1.6	0.4
Zanichella palustris	0.3	0.07
Any species above	95.6	

TABLE 7. TOTAL RAINFALL AND MAXIMUM RAINFALL FOR A DAY FOR WAUPACA

Month	1972 Total	1972 Maximum Daily Rainfall	1973 Total	1973 Maximum Daily Rainfall	1977 Total	1977 Maximum Daily Rainfall
April	1.22	0.63	4.76	0.88	3.20	0.79
May	1.66	0.65	9.18	2.33	3.22	1.40
June	1.62	1.02	1.86	0.56	4.76	1.76
July	2.33	0.55	1.81	--	3.66	1.14
August	5.19	2.00	1.91	0.46	2.90	1.89
September	7.66	2.06	3.10	--	4.47	1.84
October	2.27	1.01	2.76	1.20	2.64	1.41

TABLE 8. AVERAGE CONCENTRATIONS OF SOME CHEMICAL PARAMETERS FOR SHADOW LAKE

SHADOW LAKE
October 17, 1976 - October 25, 1977
(mg/l)

Depth	Tot-P	DRP	Org-P	Tot-N	Inorg-N	Org-N	NH_4-N	NO_3 + NO_2-N	(μmho/cm) Cond.
0	.030	.005	.025	.801	.165	.636	.087	.078	376
2	.031	.005	.026	.806	.155	.651	.086	.069	378
4	.026	.004	.022	.722	.159	.563	.077	.082	370
6	.034	.007	.027	.869	.222	.647	.101	.121	428
8	.065	.008	.057	1.225	.357	.868	.272	.085	439
10	.132	.055	.077	2.347	1.454	.893	1.412	.042	453
12	.292	.181	.111	4.862	3.883	.979	2.877	.006	476

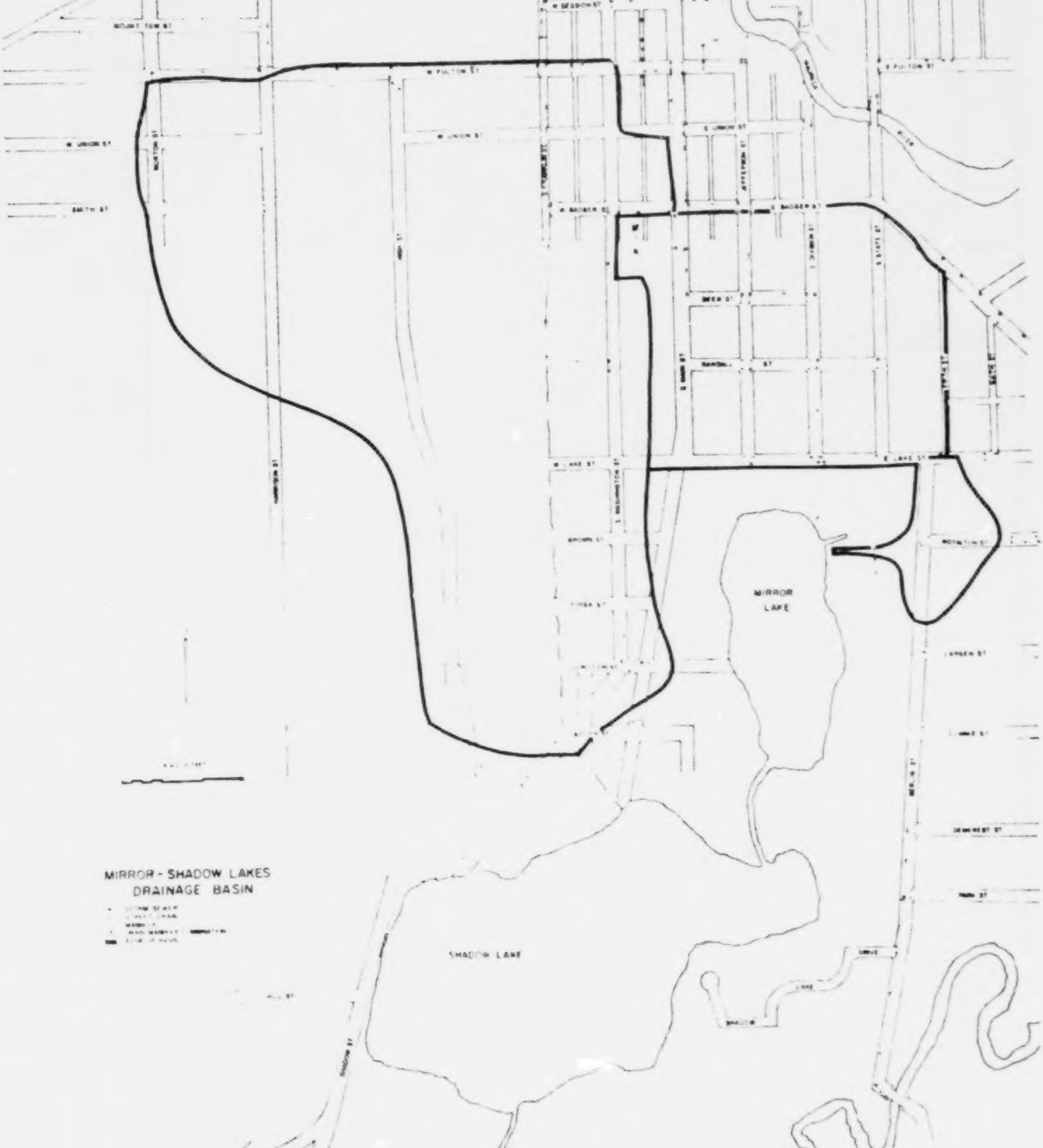

MIRROR - SHADOW LAKES
DRAINAGE BASIN
W SESSION ST
W FULTON ST
E FULTON ST
W UNION ST
E UNION ST
W BARBER ST
E BARBER ST
W LAKE ST
E LAKE ST
MIRROR
LAKE
SHADOW LAKE

Figure 1. Storm sewer drainage system to Mirror and Shadow lakes before diversion.

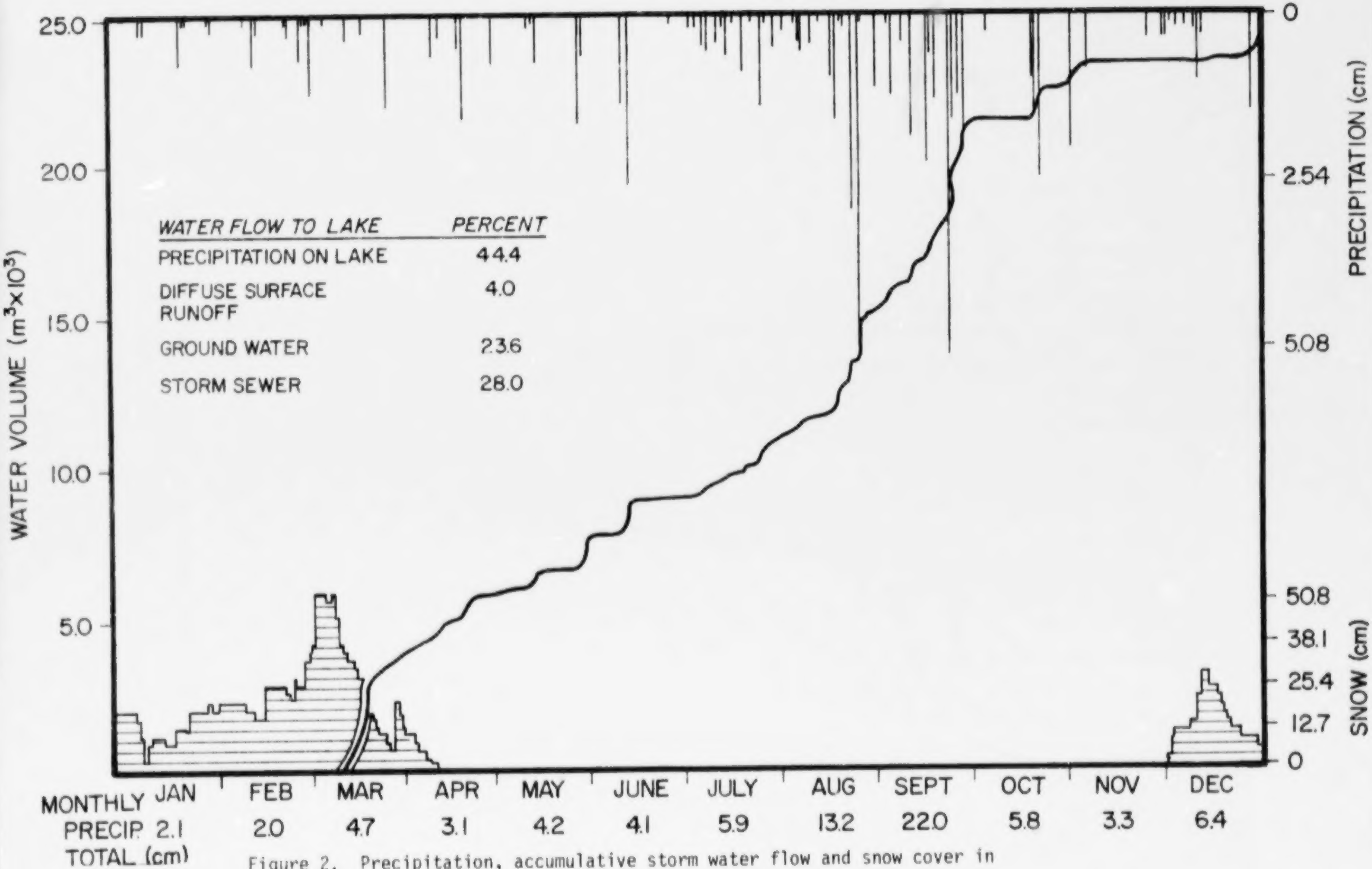


Figure 2. Precipitation, accumulative storm water flow and snow cover in Mirror Lake basin for 1972.

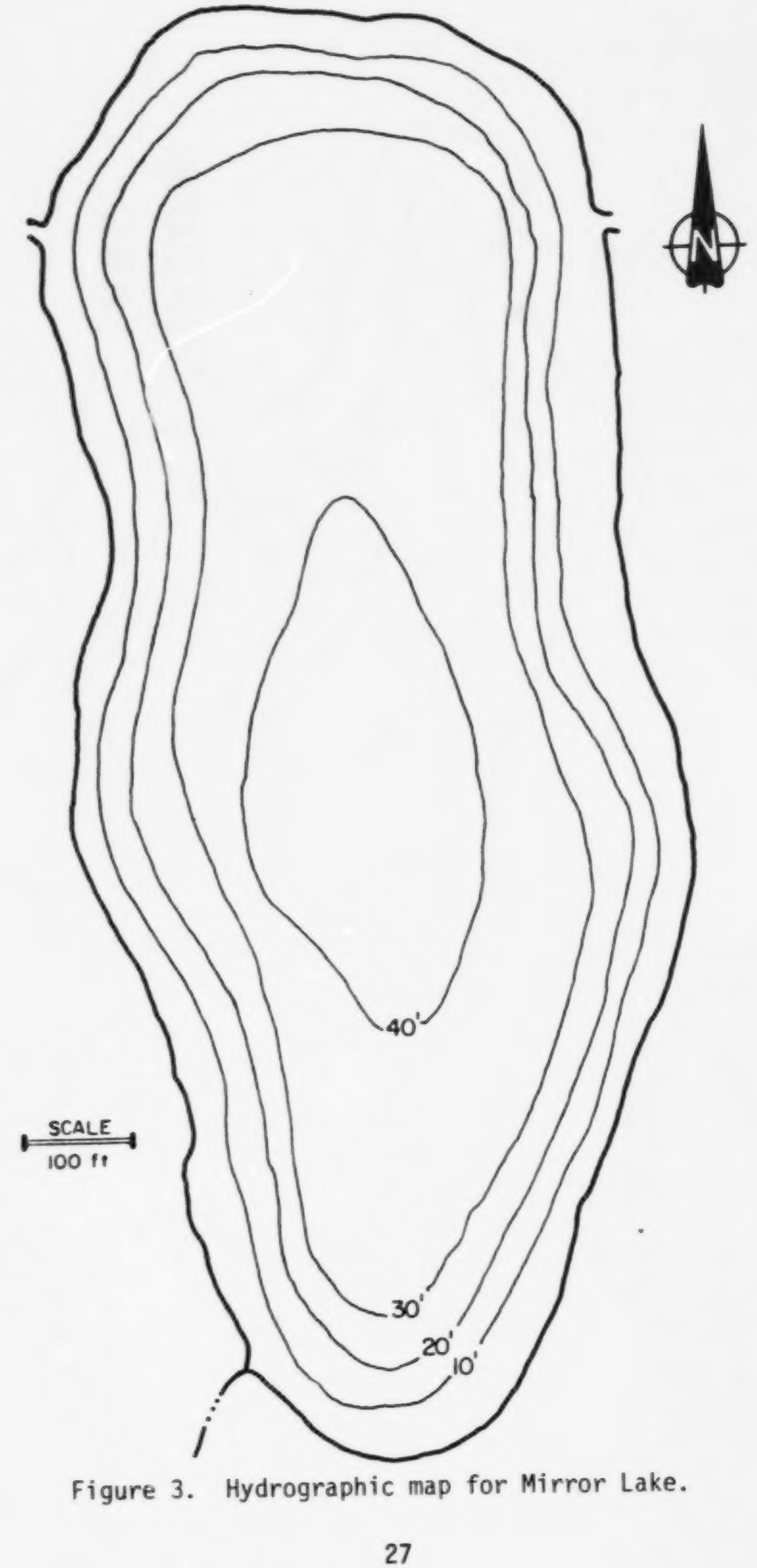
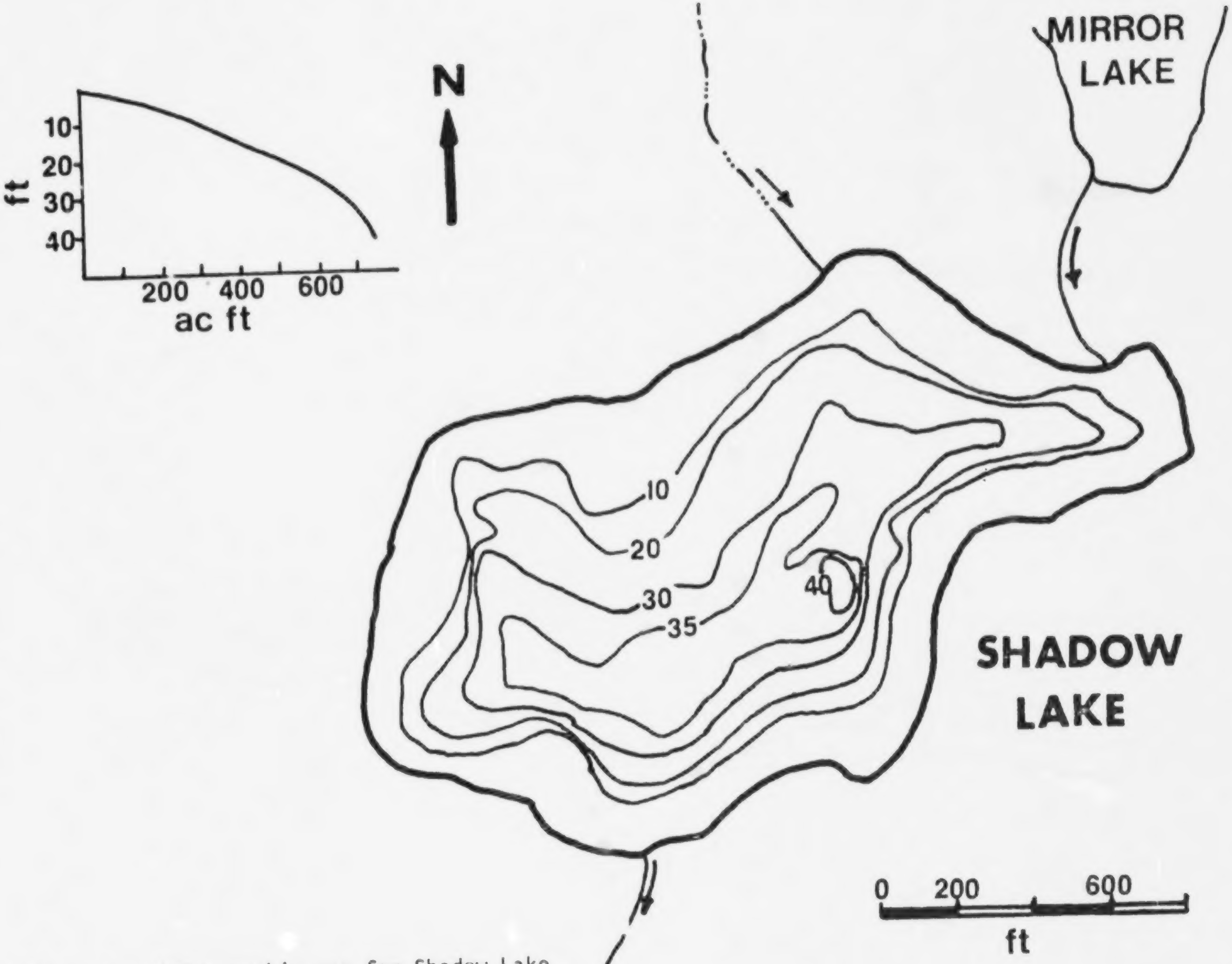

SCALE
100 ft
N
40'
30'
20'
10'

Figure 3. Hydrographic map for Mirror Lake.



MIRROR LAKE
SHADOW LAKE
N
10
20
30
35
40
0
200
600
ft
ft
10
20
30
40
200
400
600
ac ft

Figure 4. Hydrographic map for Shadow Lake.

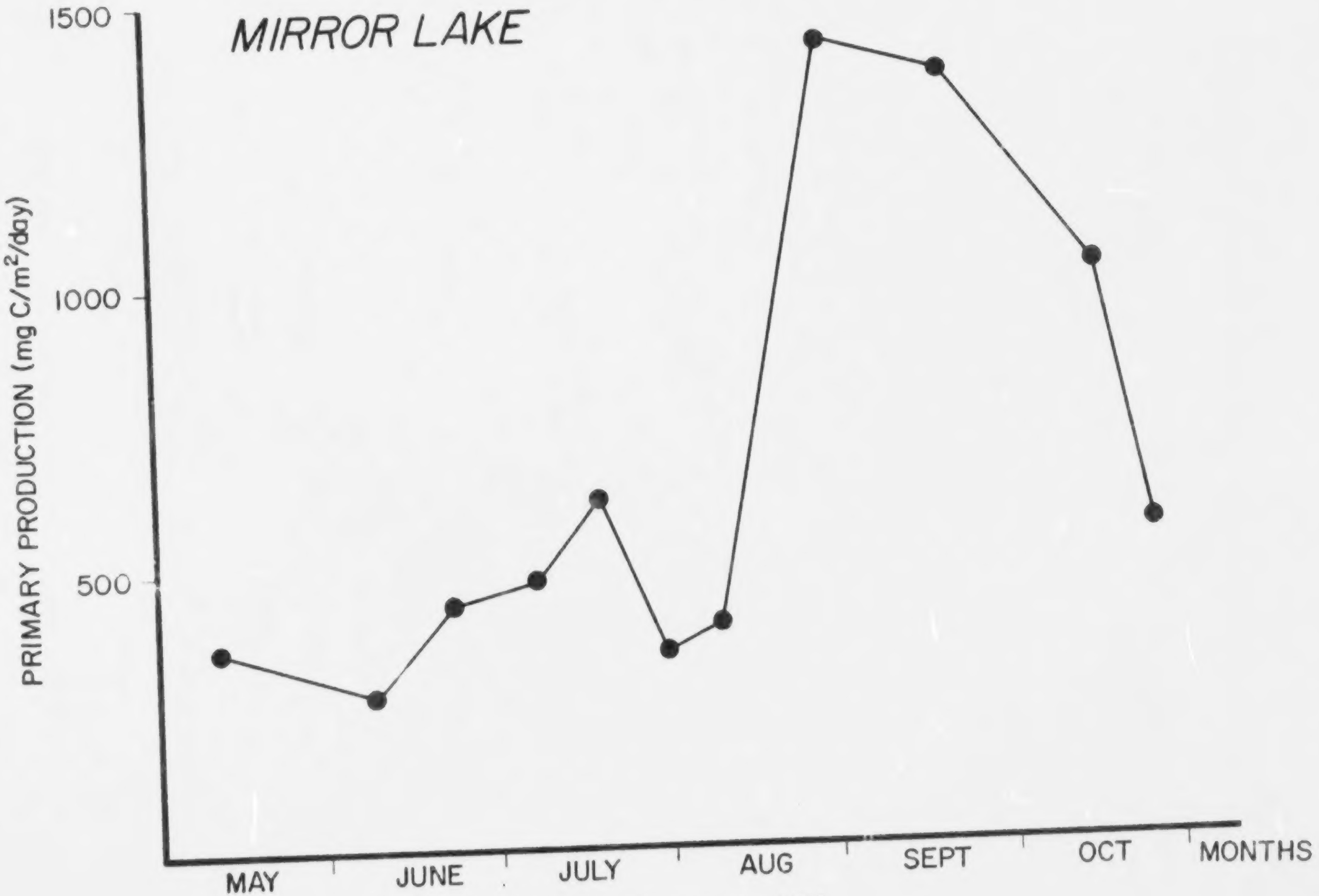


Figure 5. Primary productivity for Mirror Lake 1972.

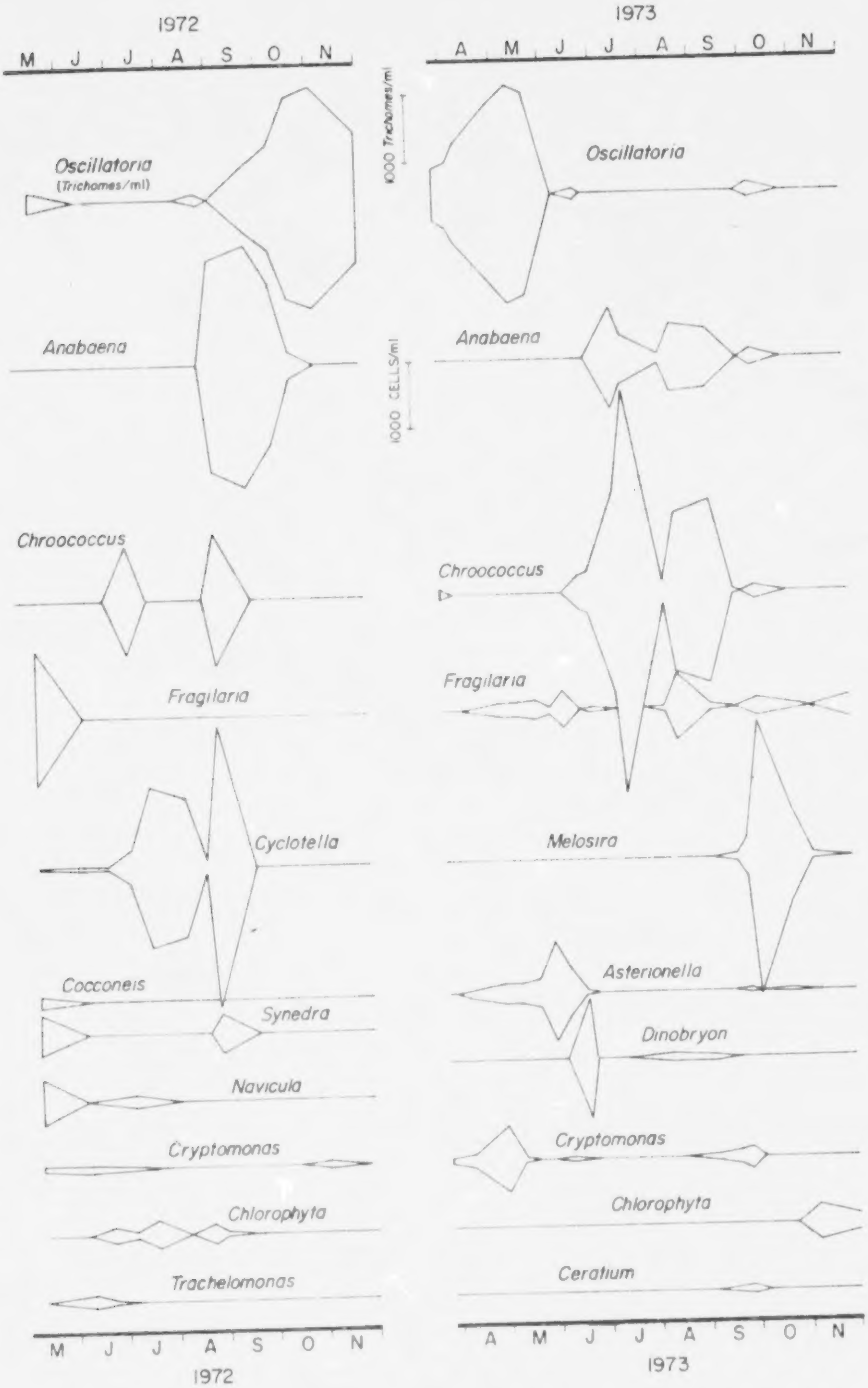


Figure 6. Phytoplankton temporal distribution for Mirror Lake 1972 and 1973.

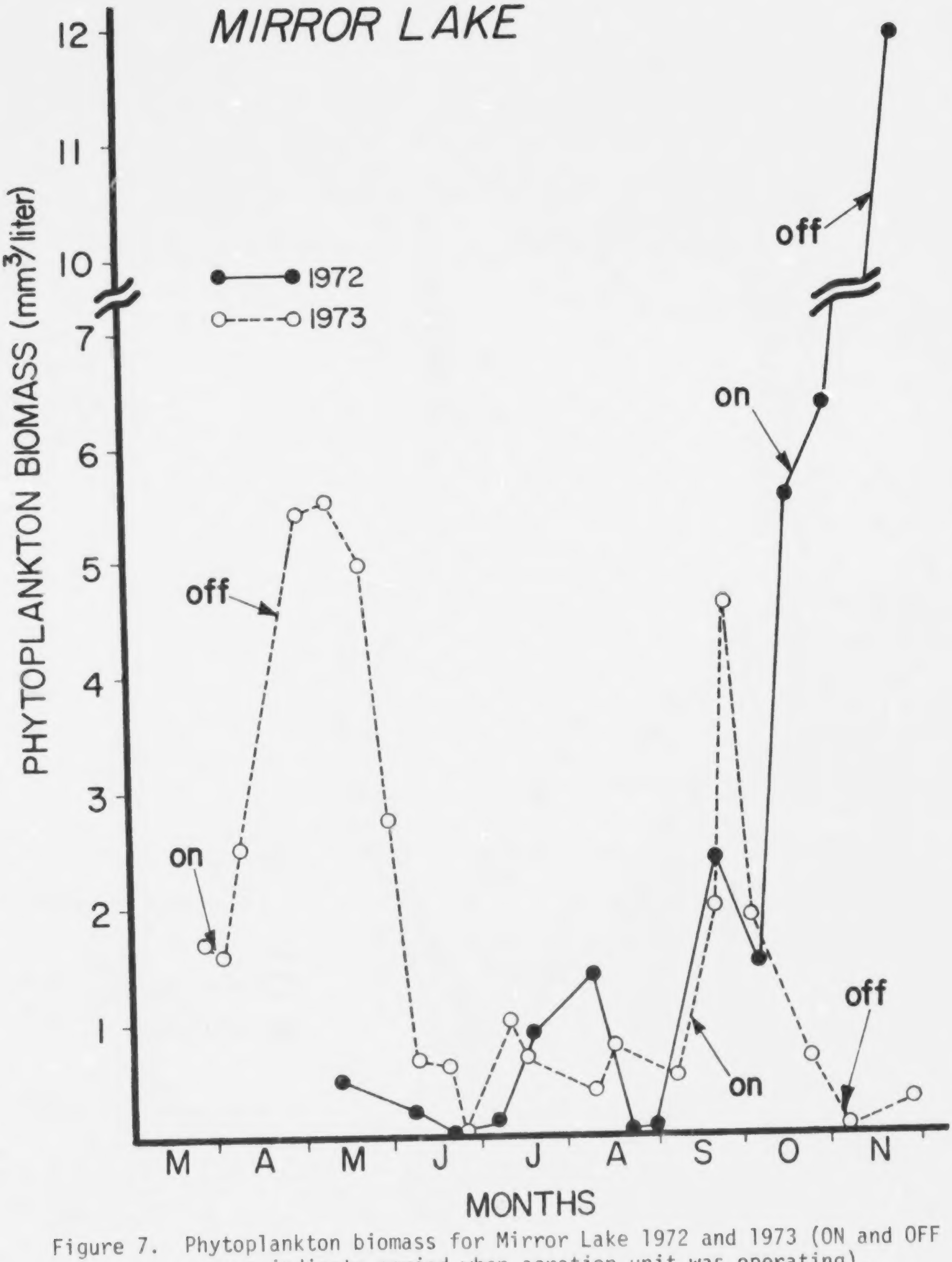


Figure 7. Phytoplankton biomass for Mirror Lake 1972 and 1973 (ON and OFF arrows indicate period when aeration unit was operating).

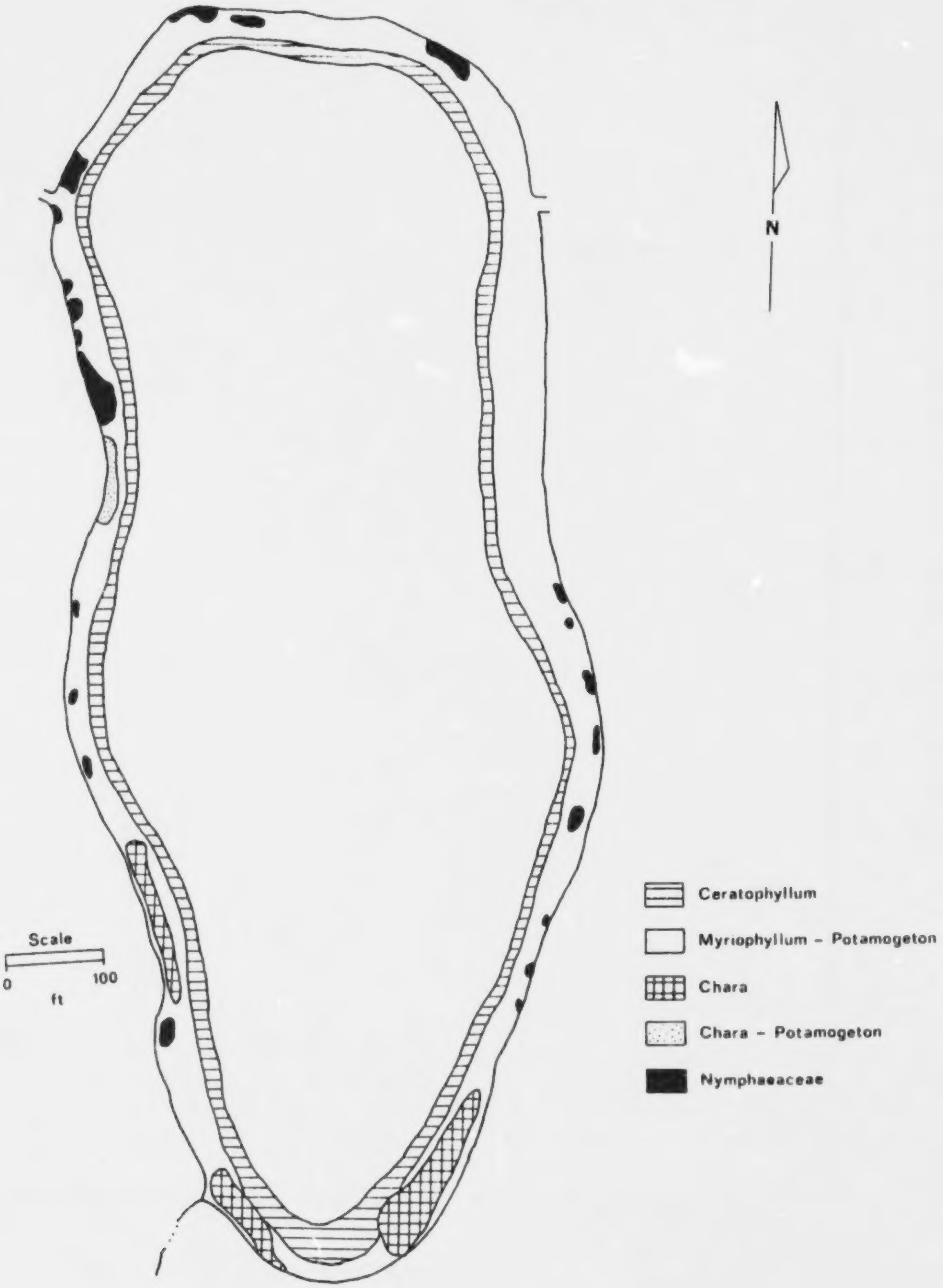

N
Scale
0
100
ft
Ceratophyllum
Myriophyllum - Potamogeton
Chara
Chara - Potamogeton
Nymphaeaceae

Figure 8. Map of Mirror Lake illustrating the major plant communities.

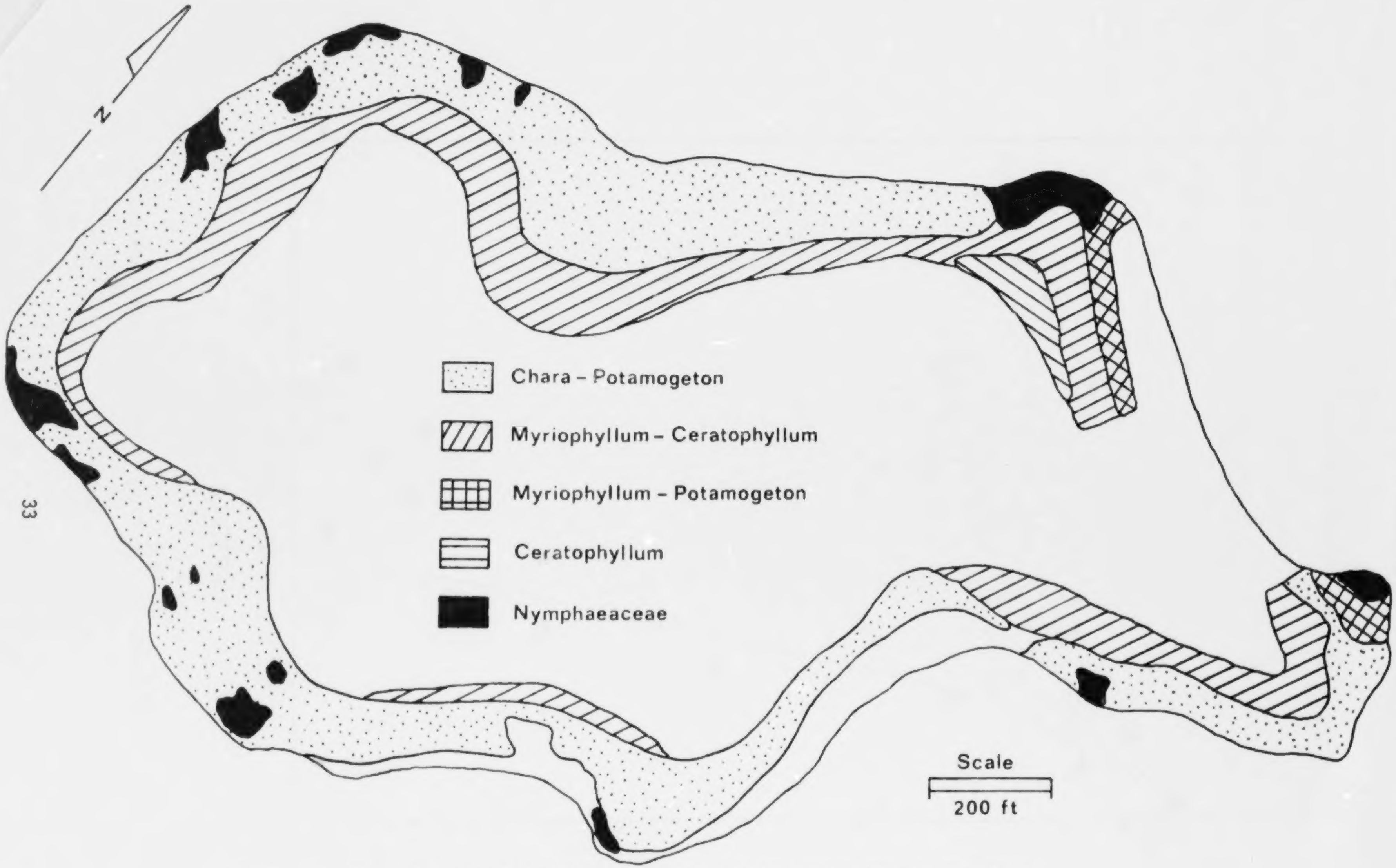

N
Chara - Potamogeton
Myriophyllum - Ceratophyllum
Myriophyllum - Potamogeton
Ceratophyllum
Nymphaeaceae
Scale
200 ft

Figure 9. Map of Shadow Lake illustrating the major macrophyte communities.

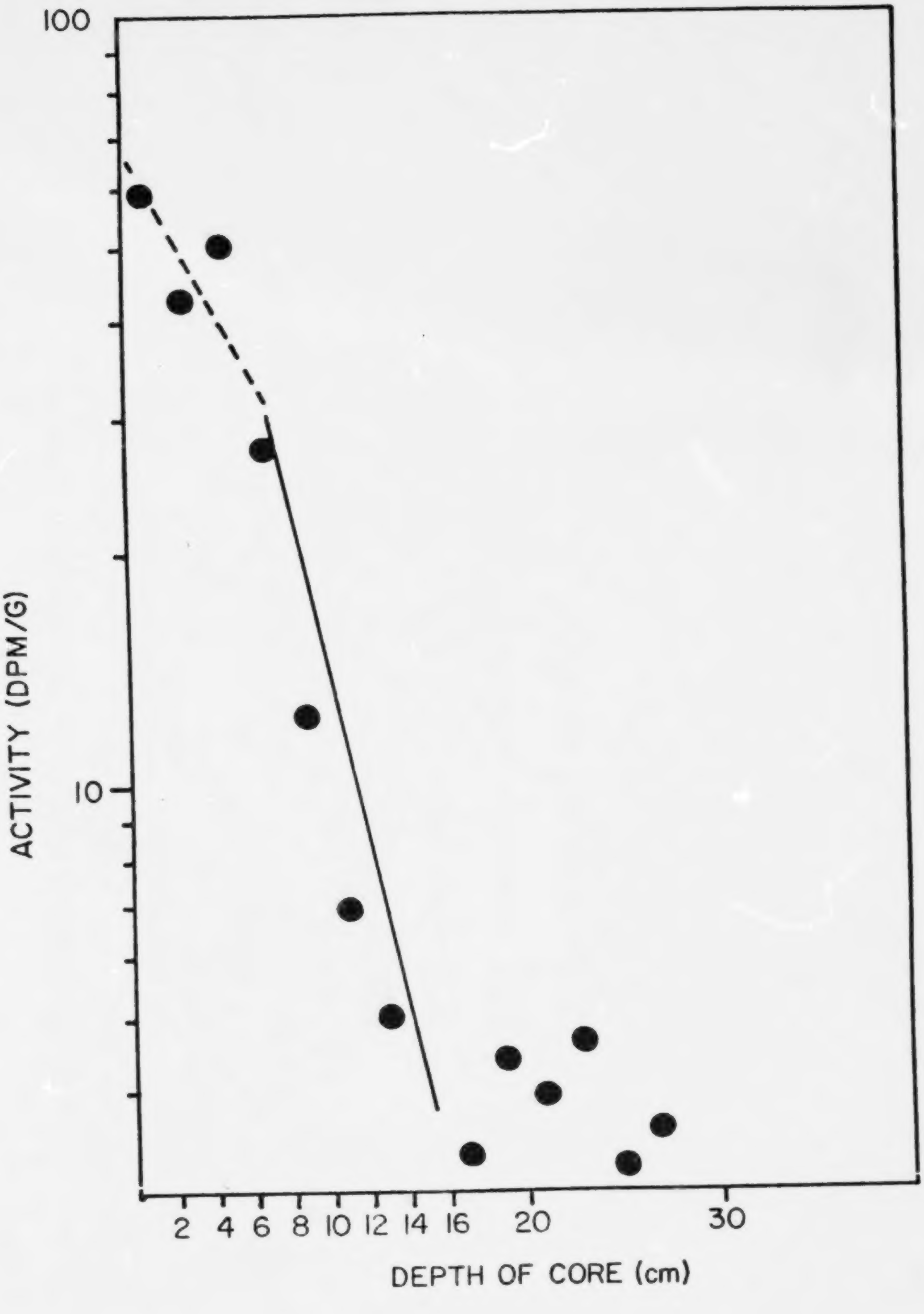


Figure 10. Radiometric dating, ^{210}Pb, of a sediment core from Mirror Lake.

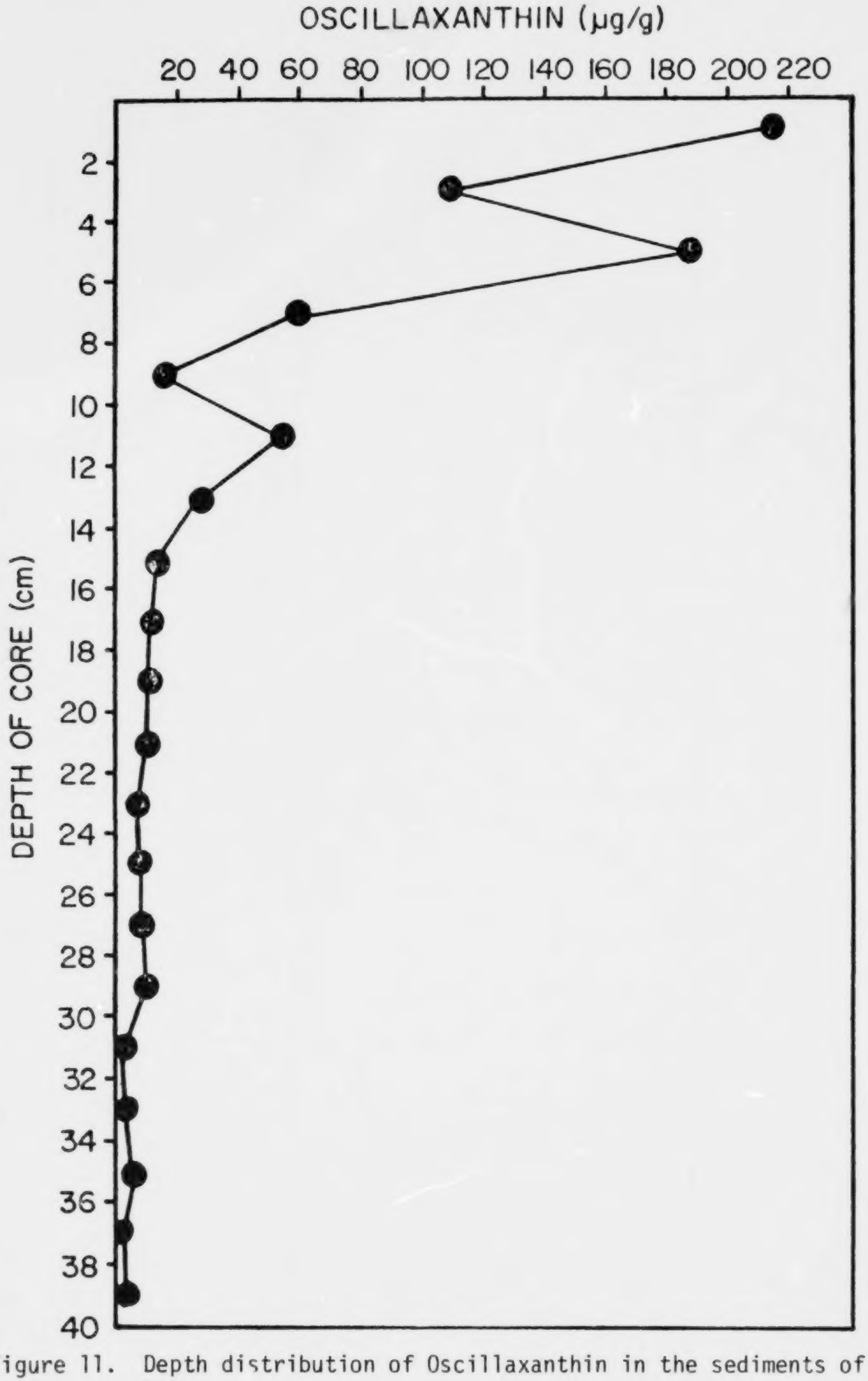


Figure 11. Depth distribution of Oscillaxanthin in the sediments of Mirror Lake.

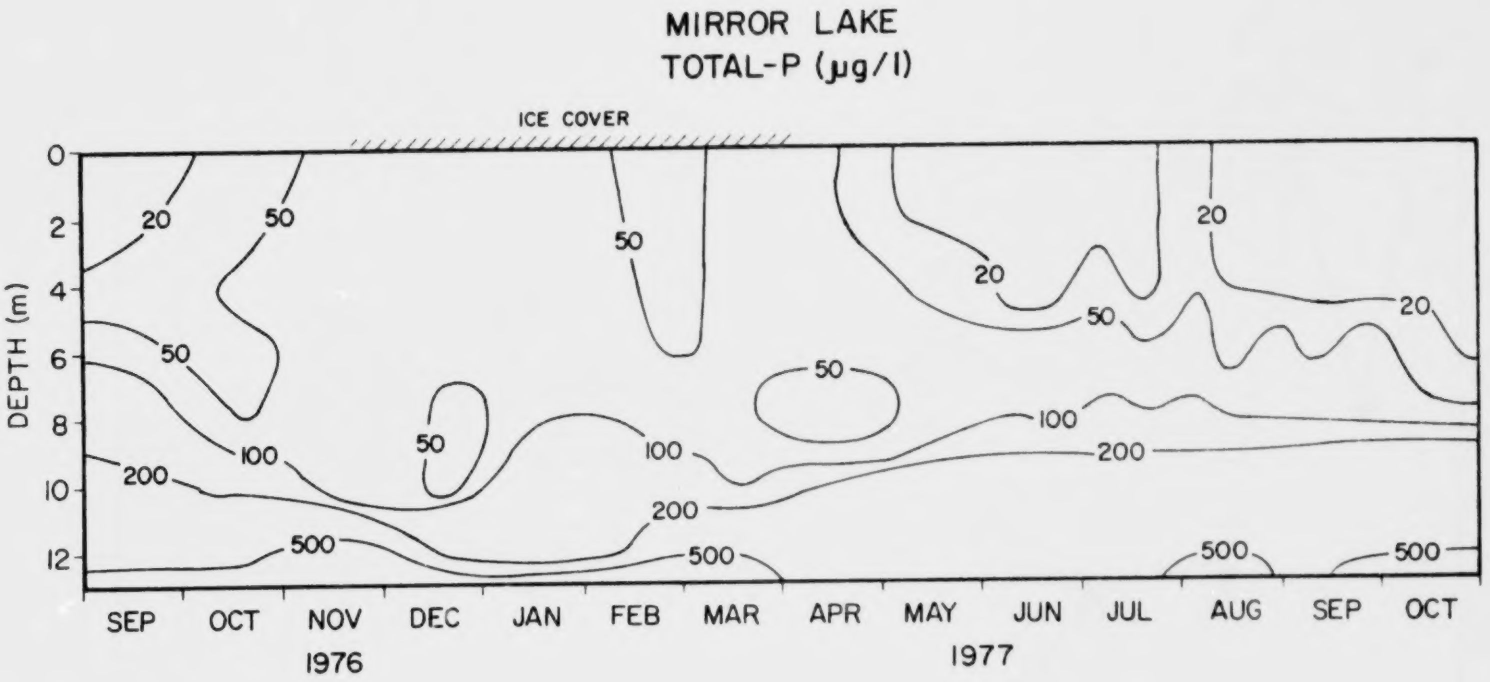


Figure 12. Total phosphorus isopleths for Mirror Lake from September 1976 through October 1977.

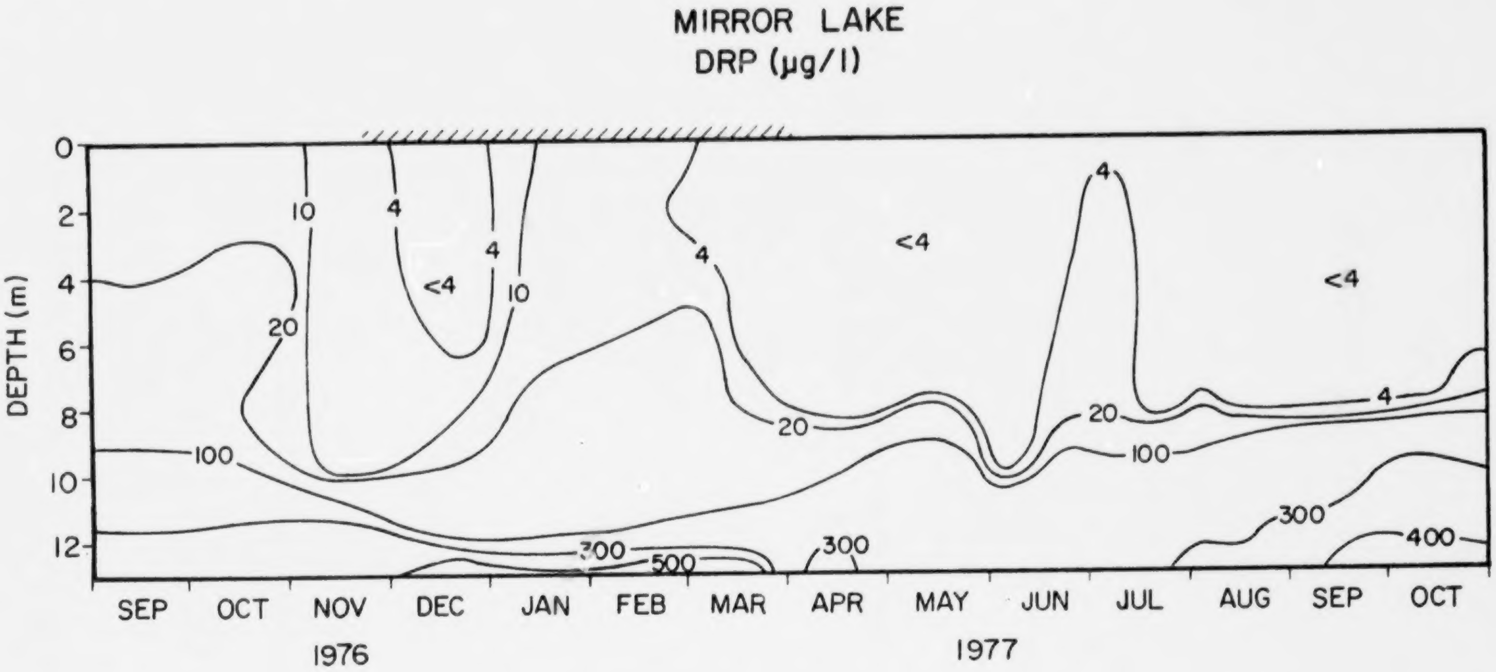


Figure 13. Dissolved reactive phosphorus (DRP) isopleths for Mirror Lake from September 1976 through October 1977.

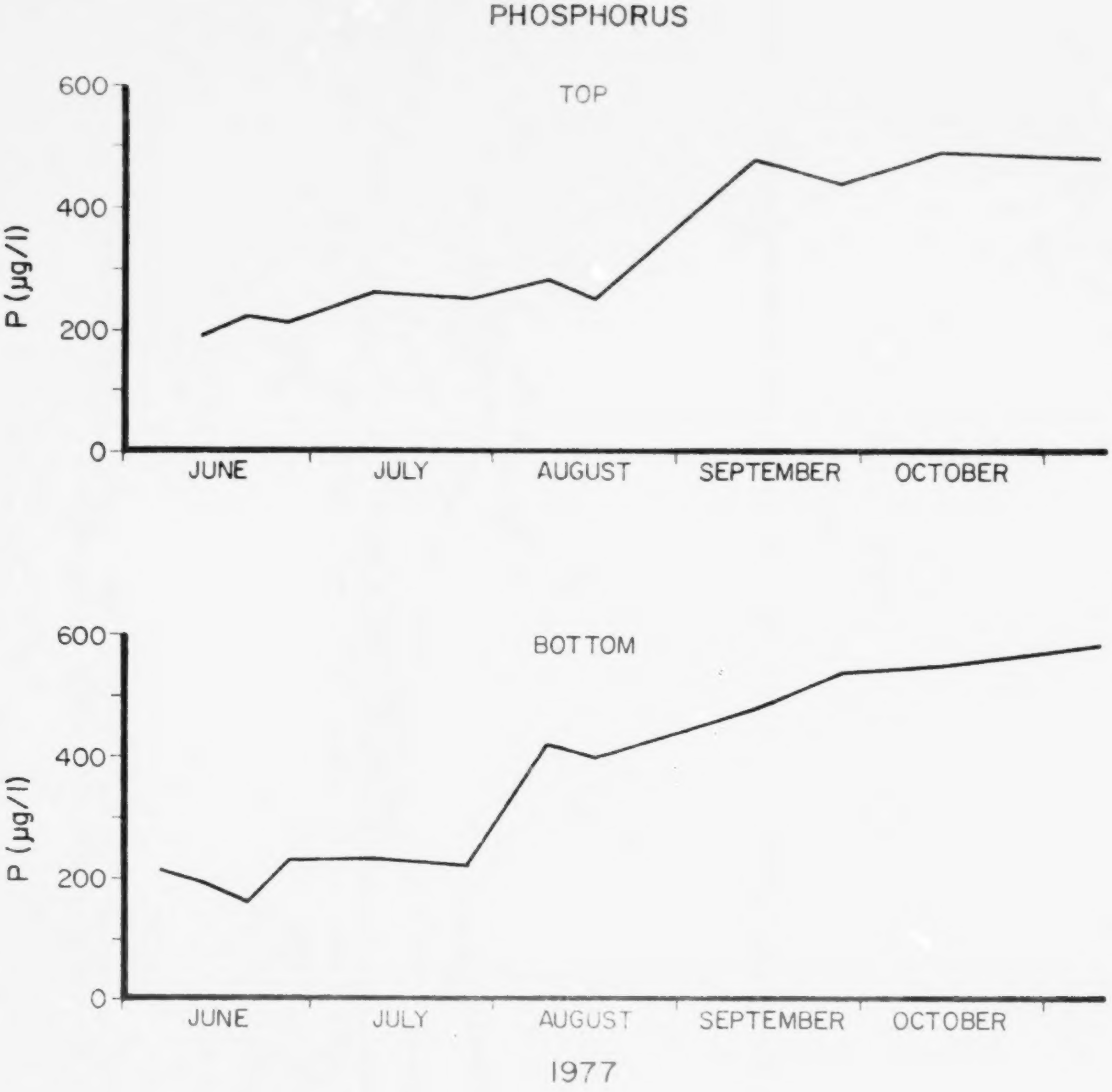


Figure 14. Total phosphorus trends in the nutrient regeneration chambers.

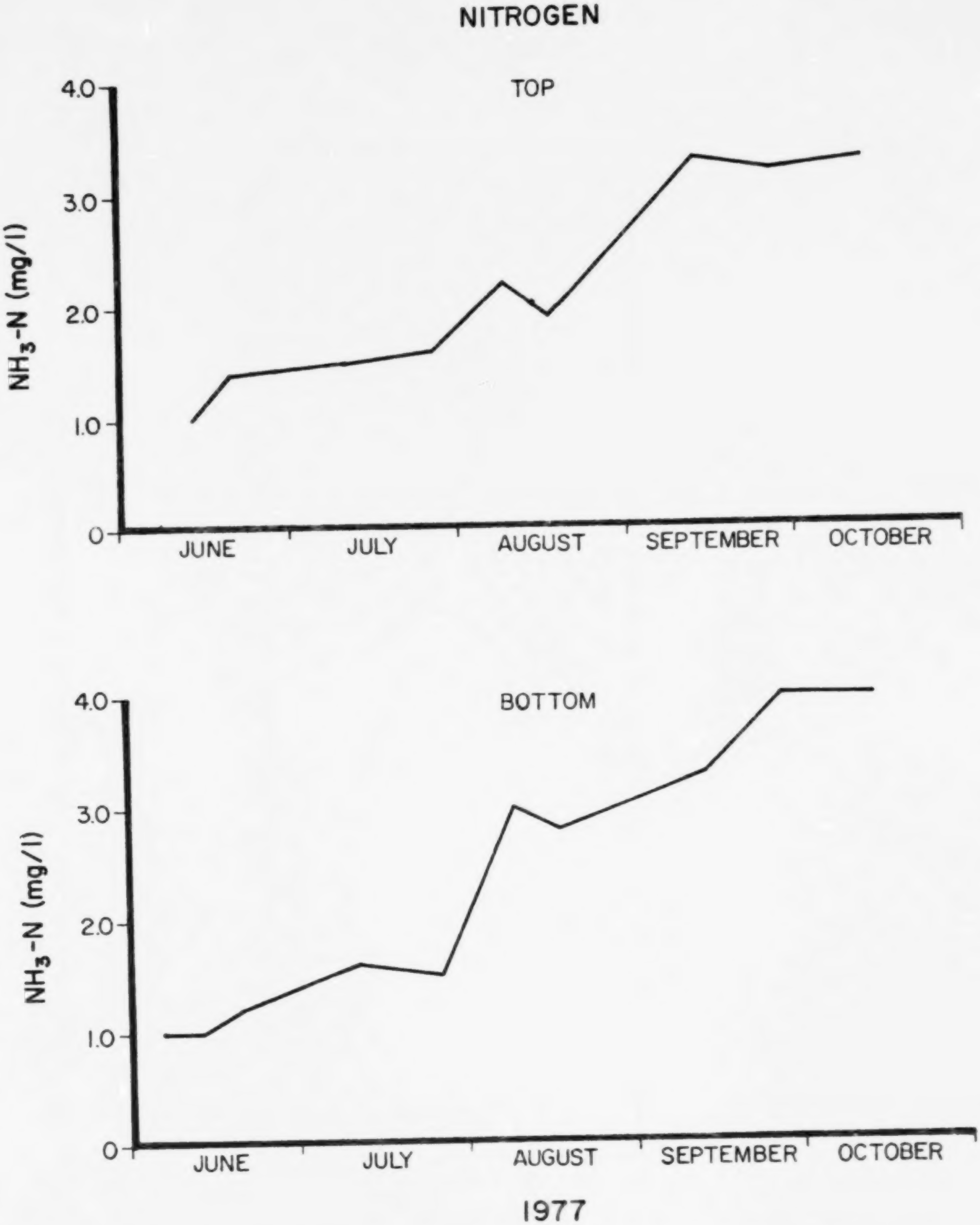


Figure 15. Ammonium-nitrogen trends in the nutrient regeneration chambers.

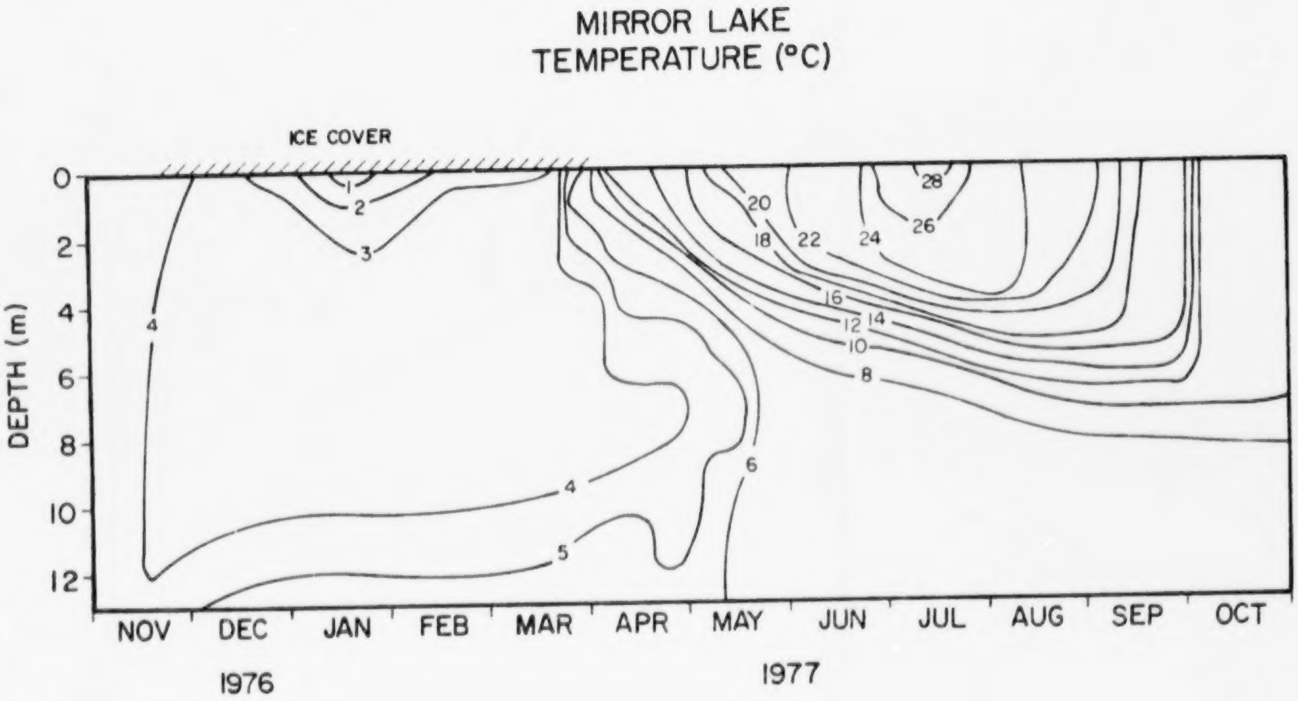


Figure 16. Temperature isopleths for Mirror Lake from November 1976 through October 1977.

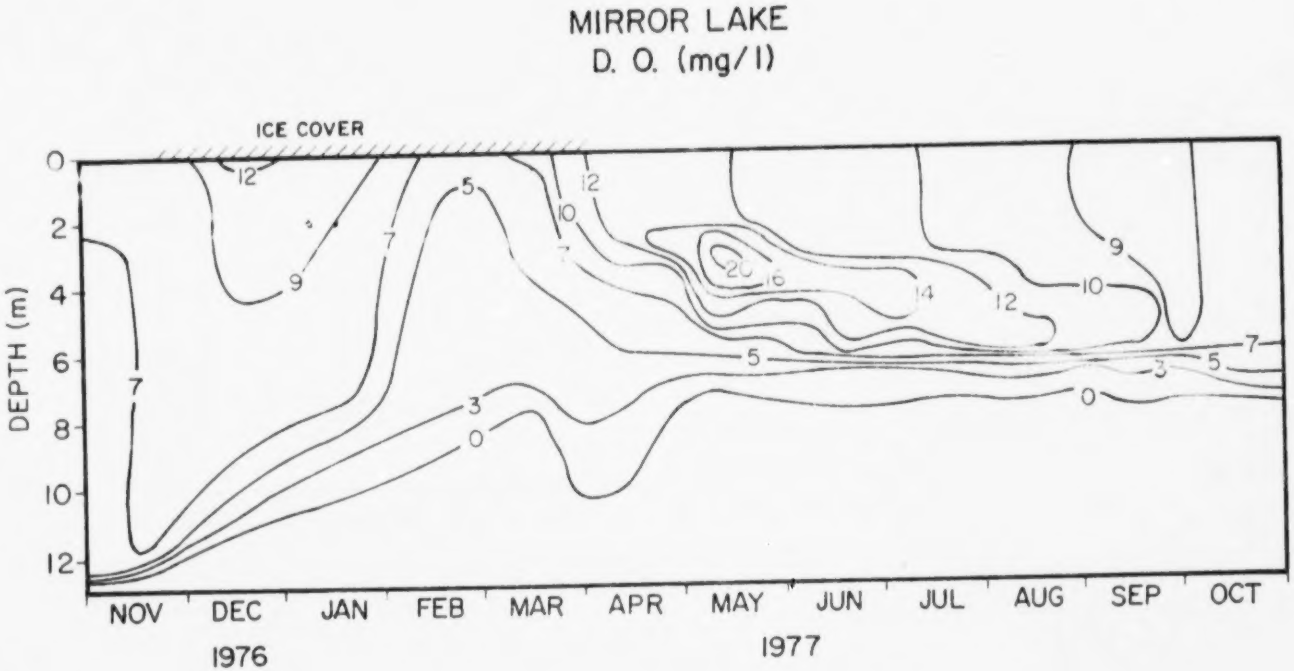


Figure 17. Dissolved oxygen (DO) isopleths for Mirror Lake from November 1976 through October 1977.

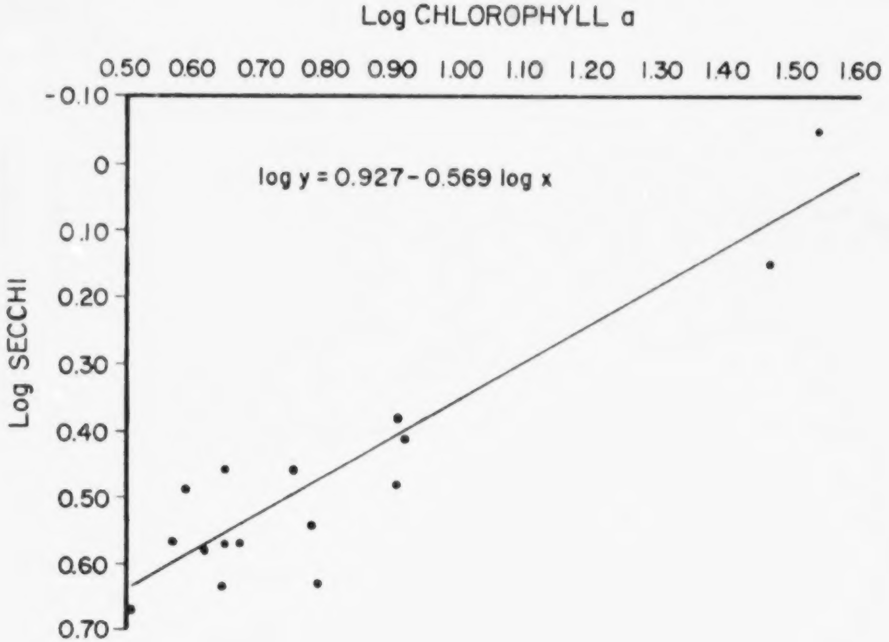


Figure 18. Log chlorophyll a vs log Secchi disc depth for Mirror Lake, April 1977 through October 1977 (r=0.92).

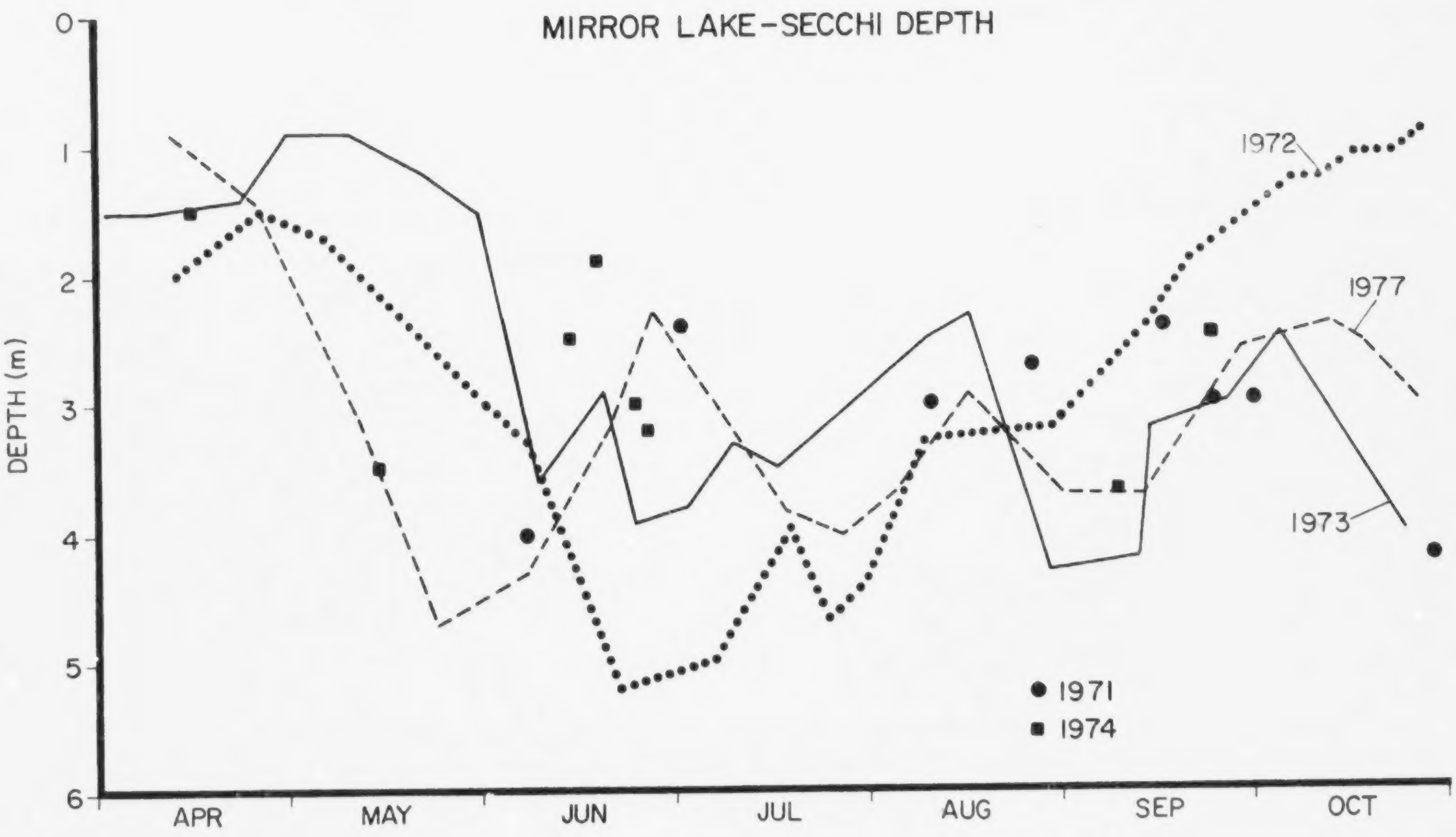


Figure 19. Comparison of Secchi disc depths for Mirror Lake before and after storm sewer diversion.

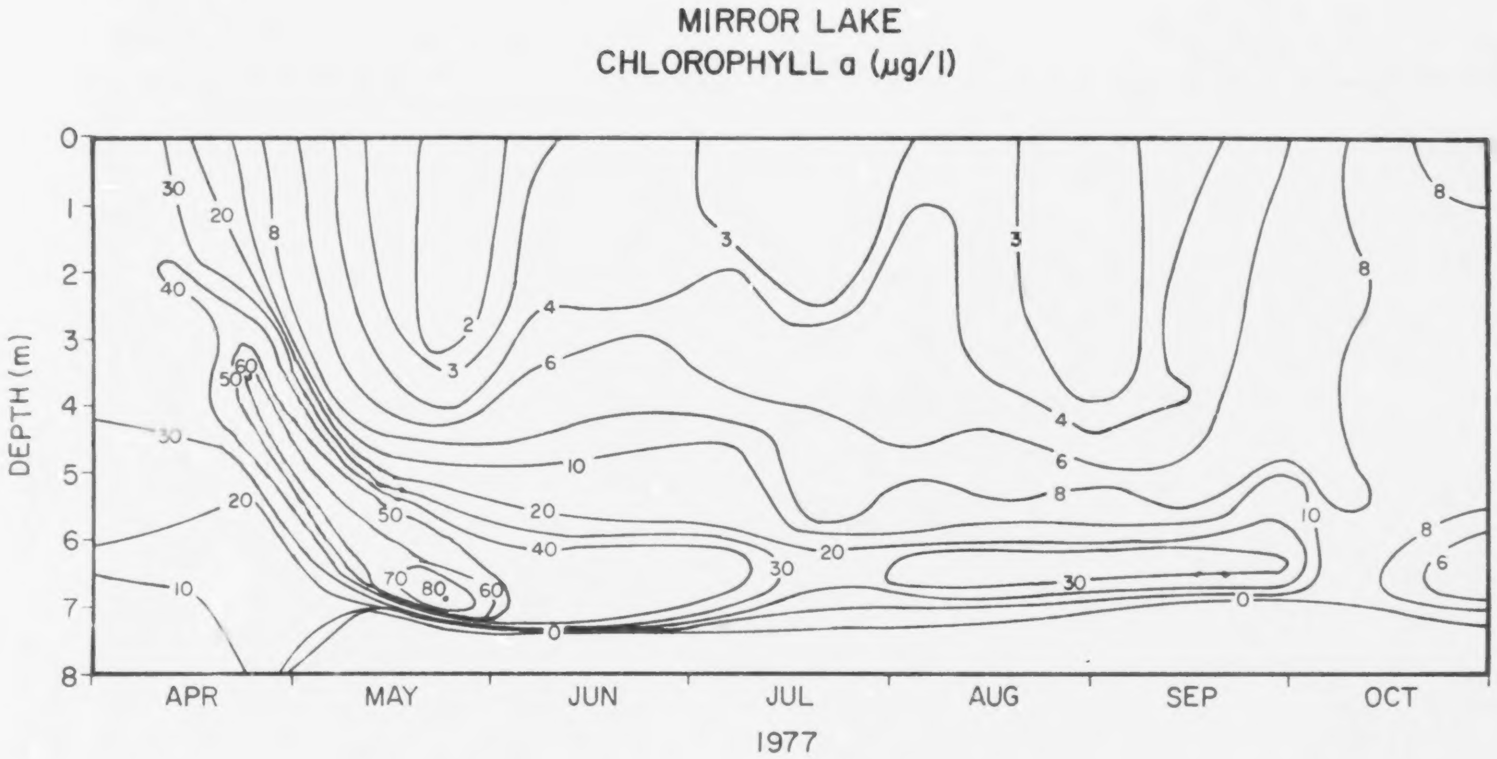


Figure 20. Chlorophyll a isopleths for Mirror Lake after storm sewer diversion.

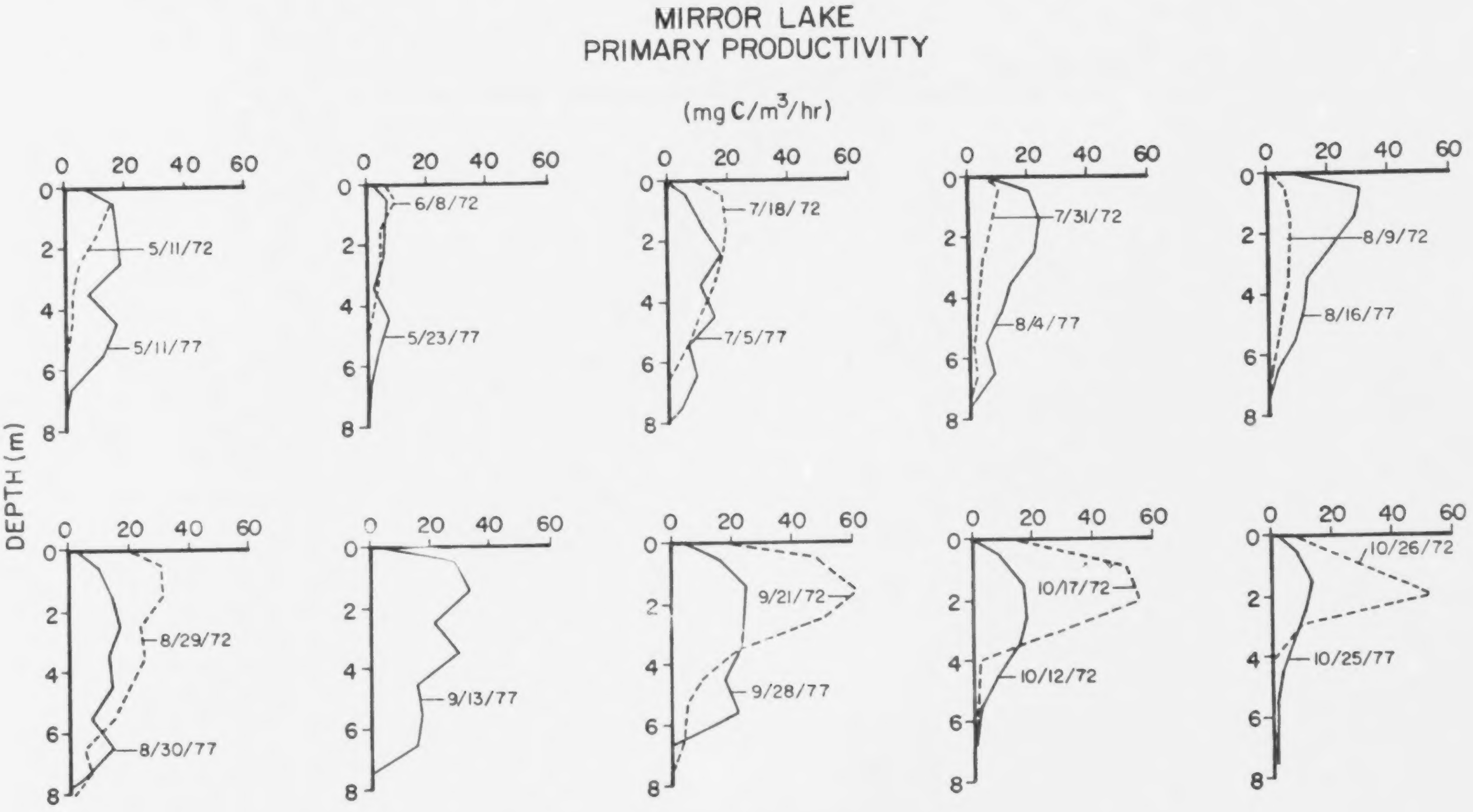


Figure 21. Comparison of selected primary productivity profiles before and after storm sewer diversion.

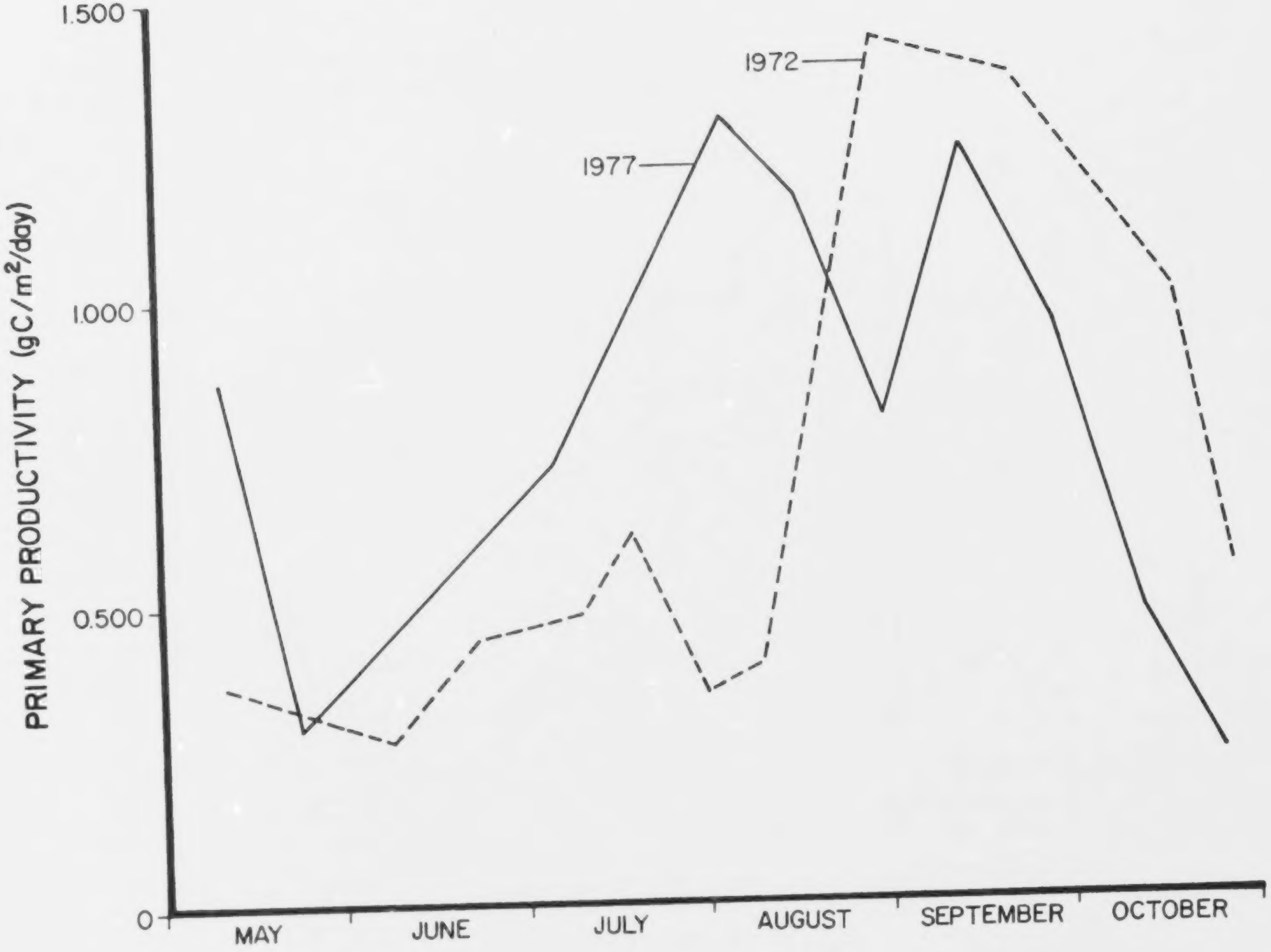


Figure 22. Comparison of primary productivity measured on an areal basis before and after storm sewer diversion.

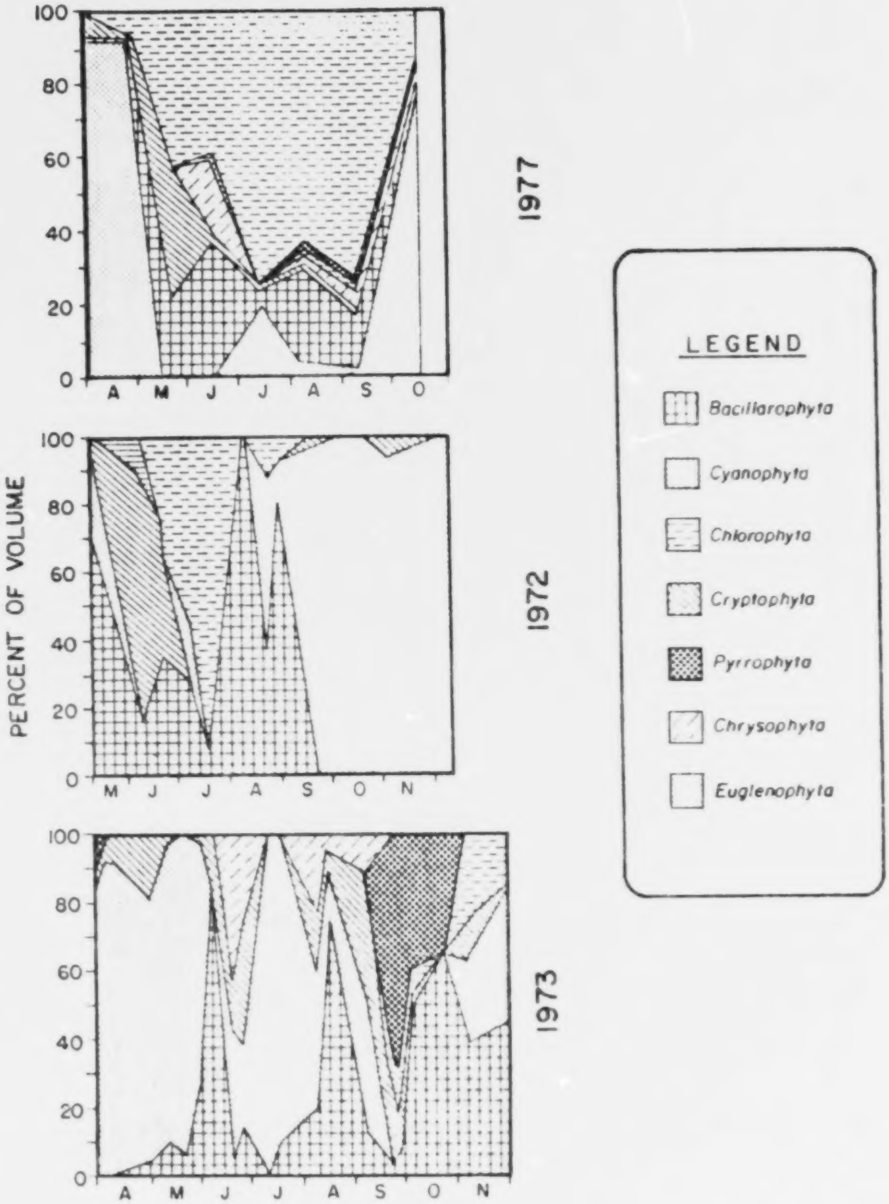


Figure 23. Comparison of phytoplankton species composition as per cent biomass before and after storm sewer diversion.

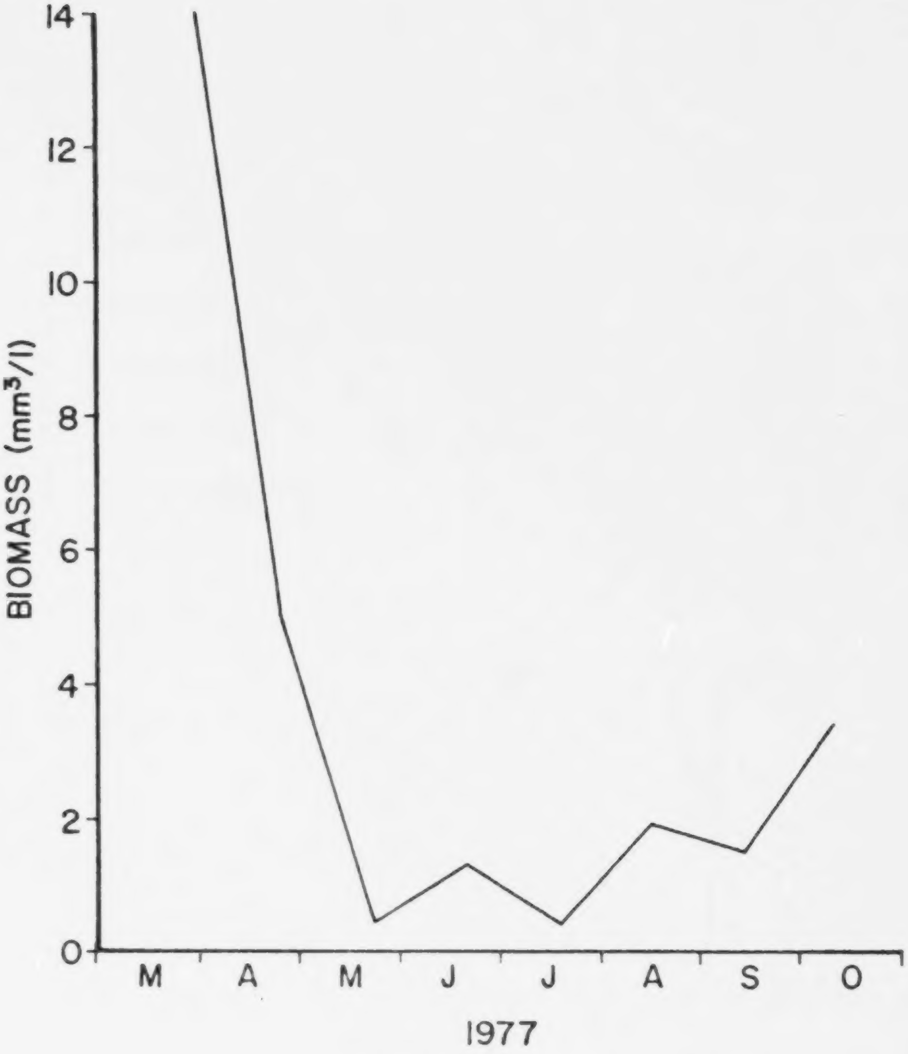


Figure 24. Total phytoplankton biomass after storm sewer diversion.

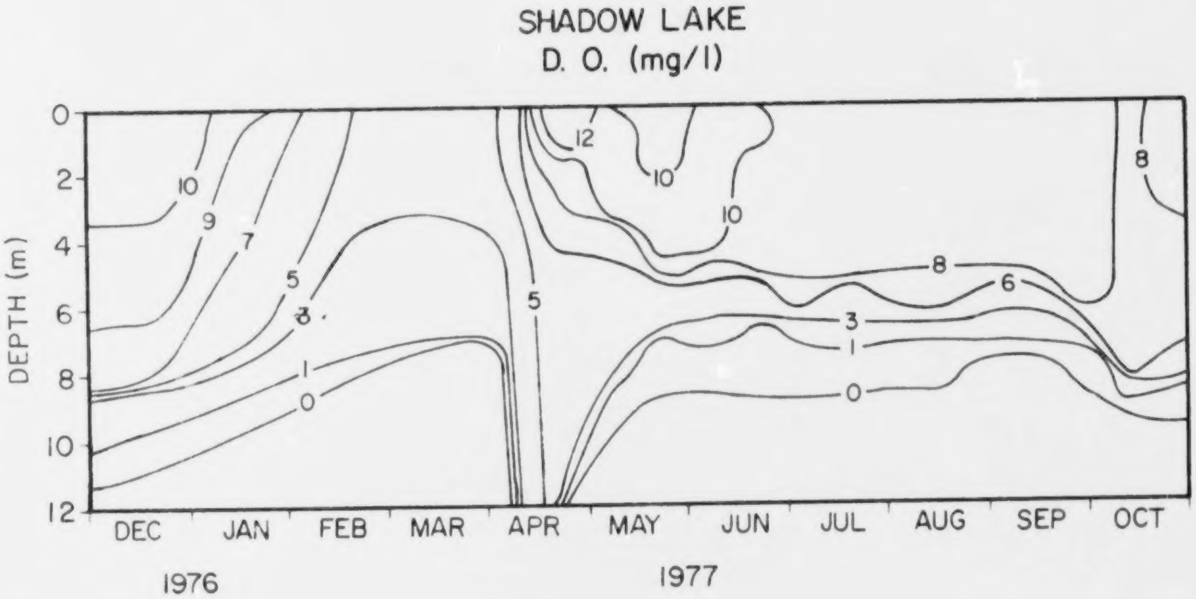


Figure 25. Dissolved oxygen (DO) isopleths for Shadow Lake from December 1976 through October 1977.

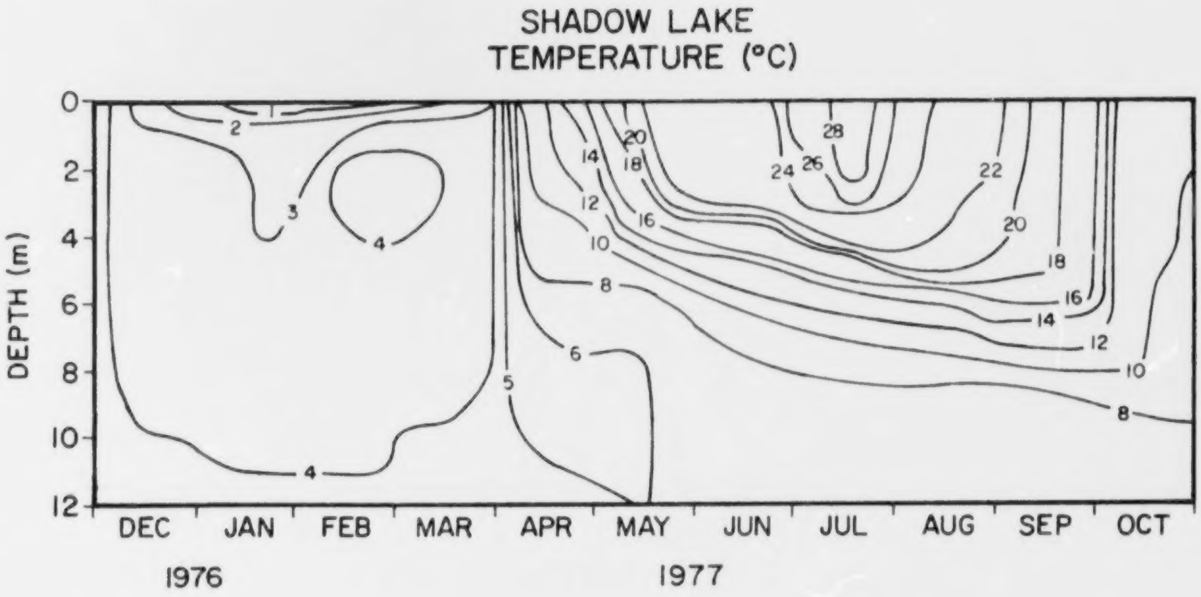


Figure 26. Temperature isopleths for Shadow Lake from December 1976 through October 1977.

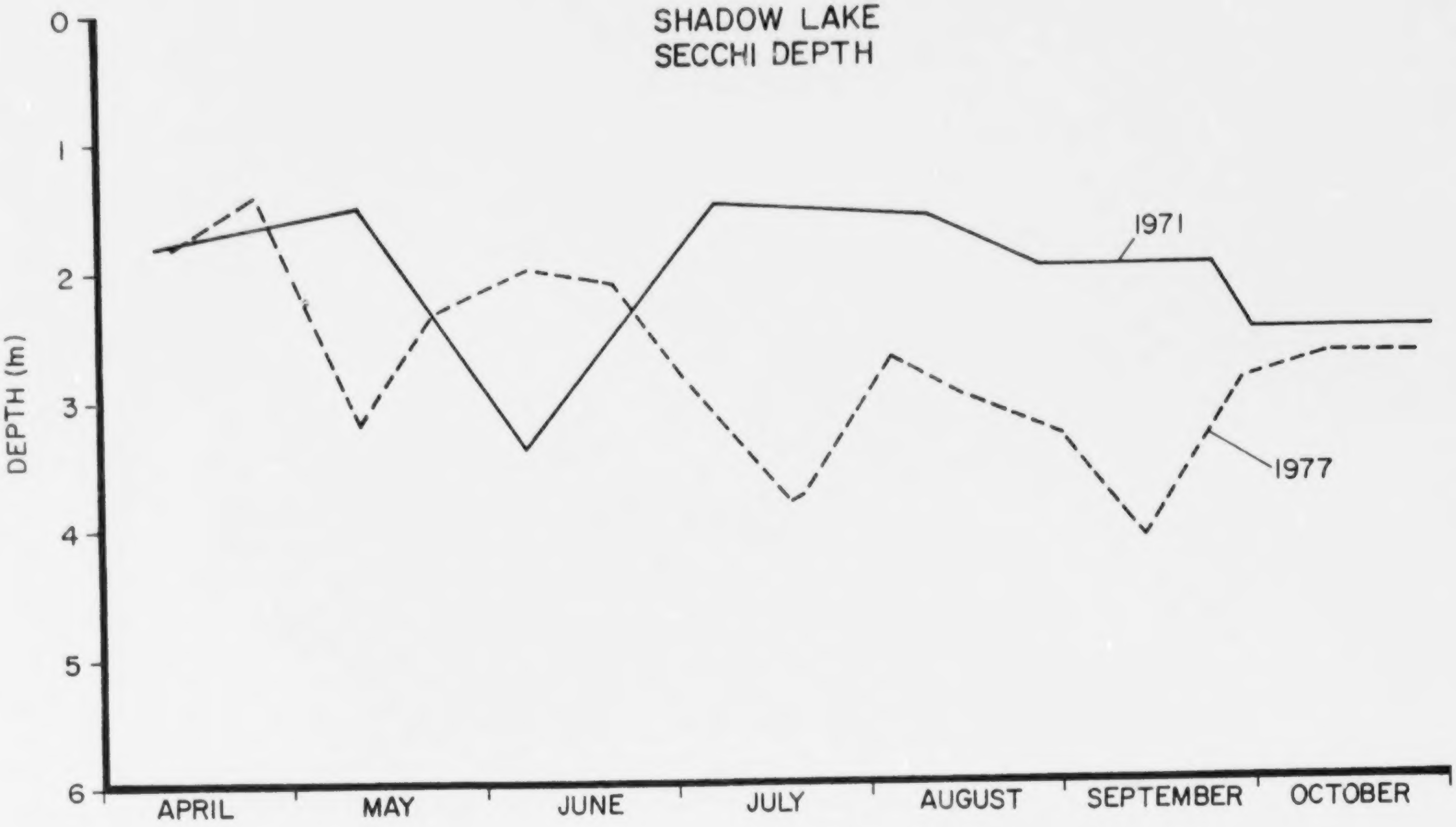


Figure 27. Comparison of Secchi disc depths for Shadow Lake before and after storm sewer diversion.

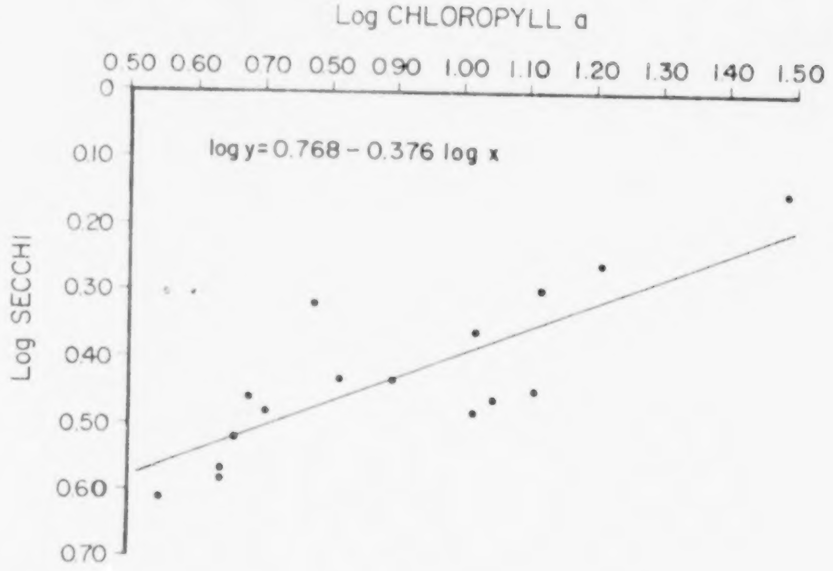


Figure 28. Log chlorophyll a vs log Secchi disc depth for Shadow Lake, April 1977 through October 1977 (r = 0.78).

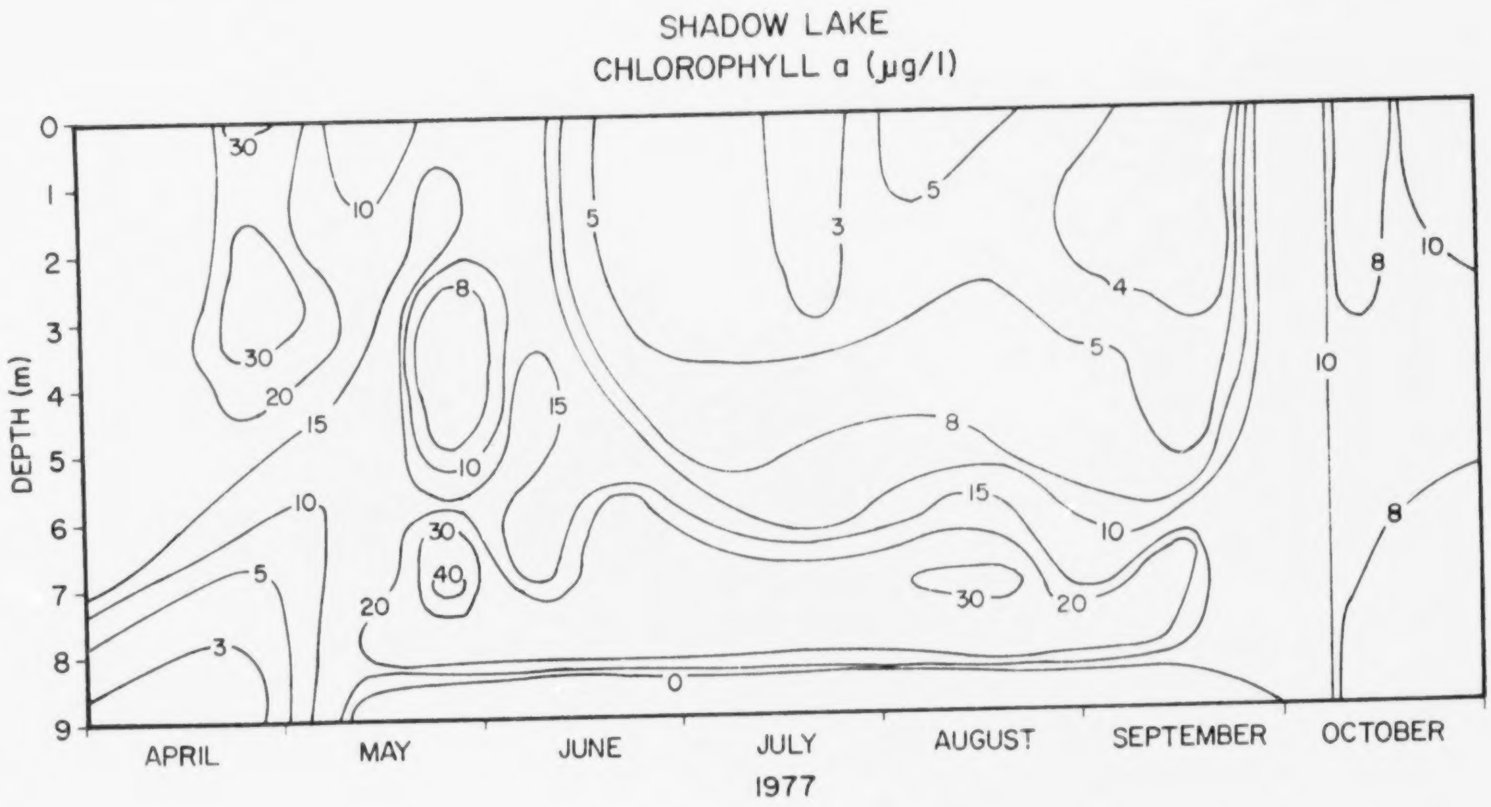


Figure 29. Chlorophyll a isopleths for Shadow Lake after storm sewer diversion.

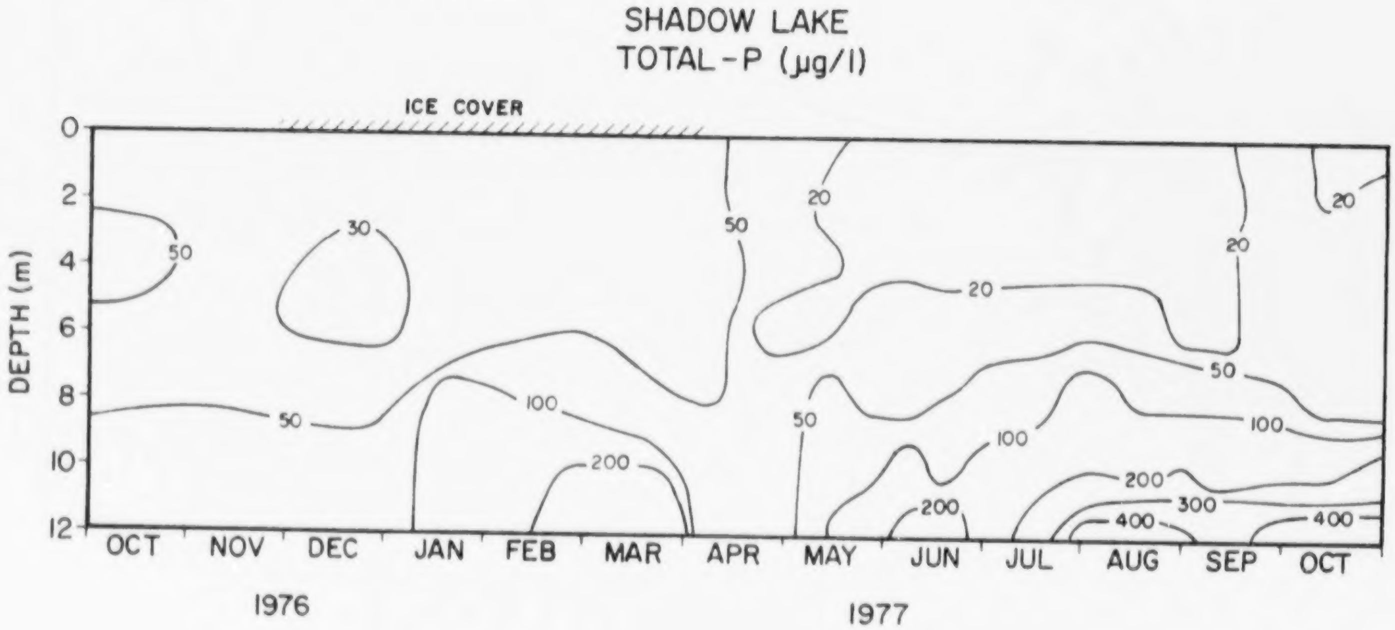


Figure 30. Total phosphorus isopleths for Shadow Lake from October 1976 through October 1977.

THE WHITE CLAY MANAGEMENT PLAN - DEVELOPMENT, IMPLEMENTATION AND EVALUATION

by

J. Peterson[1] and F. Madison[2]

Enactment of the Amendments to the Federal Water Pollution Control Act of 1972 (Public Law 92-500) resulted in changing the primary emphasis of the nation's efforts to control the pollution of its waters. Instead of dealing with levels of pollutants in receiving water, Public Law 92-500 directed that pollutants be controlled at their source, whether that source is a treatment plant outflow pipe, an urban storm drain, a farmer's field, or a construction site. Particular attention was focused on the agricultural community as questions were raised concerning the amounts of nutrients and sediments mobilized by agricultural operations and the effects of those materials on water quality.

Generally, pollutants arising from agricultural operations are recognized as nonpoint source although in one part of the rural scene there has been a good deal of confusion. Public Law 92-500 defined animal feedlots, barnyards and rest areas as point sources of pollution and directed the Environmental Protection Agency to develop guidelines for regulating the discharge of pollutants from them. Although there has been considerable controversy about how many animals should be contained in a feedlot before a discharge permit is required, it is apparent that outflows of nutrients from these areas will need to be controlled whether they are considered to be point or nonpoint sources.

Dairy fairming dominates the 1,215 hectare White Clay Lake Watershed which is located in eastern Shawano County, Wisconsin. Water quality in the 95 hectare lake is generally good. Because there is almost no development along the shore, this project provided an excellent opportunity to study the effects of agricultural runoff on water quality and thus to increase the understanding of lakes and lake problems.

Concern for protection of the lake by the residents of the watershed led to a request for funds from the Upper Great Lakes Regional Commission (UGLRC) in 1973. A grant was subsequently awarded to the University of Wisconsin-Extension in September of that year and installation of the monitoring network commenced immediately. Additional monies were provided the following year by the same agency to support continued monitoring activities. It should be noted that in 1971 the U.S. Agriculture and Stabilization Service (ASCS) in recognition of some of the problems of the watershed made special cost-share funds available for the installation of animal waste storage facilities.

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Under the provisions of Chapter 33 of the Wisconsin Statutes, the Inland Lake Protection and Rehabilitation Act, the Town of Washington, which includes White Clay Lake, formed the White Clay Lake Protection and Rehabilitation District to ensure the future protection of the lake. Project personnel, working with residents of the watershed and personnel of the U.S. Soil Conservation Service (SCS) and the County Extension Office, developed a comprehensive management plan for the watershed. The plan included the construction and installation of measures to control sediment and nutrient movement from barnyards, feedlots, waterways and cropped areas. Using data from the UGLRC-sponsored project to meet feasibility requirements imposed by Chapter 33, the Lake District submitted an application for funds to implement their management plans. Grants totaling $214,500 were awarded for the project by the Wisconsin Department of Natural Resources (DNR) and the U.S. Environmental Protection Agency (EPA).

Of the lake protection projects submitted to EPA from all the states for funding in 1975, the White Clay Lake proposal was the only one providing lake protection solely through intensive watershed management. Construction of the first of the land management practices began in the fall of 1976, and installations were nearly completed by the end of 1977.

While White Clay Lake is considered to be of good quality now, several recent changes in agricultural practices threaten to produce adverse effects. The increase of dairy animal units in the watershed is the result of fewer, but larger herds (average about 75 to 100 cattle). Concurrently, more herds are being held on feedlots rather than on pastures, and more emphasis is being placed on production of corn with less emphasis on oats and hay in crop rotations. All of these changes tend toward greater potential for nutrient and sediment transport to the lake.

The White Clay Lake Watershed is on a gently rolling glacial till plain of Valderan age. A relatively short growing season with an average of 130 frost-free days and fairly youthful soils (classified as Alfic Haplorthods which have high carbonate content and modest amounts of expansible clays) favor dairy farming with crop rotations that include successive years of corn and oats followed by a minimum of four years of alfalfa.

Base maps have been prepared showing the SCS detailed soil survey, land elevations at 1.2 m contour intervals, the DNR bathymetry records of the lake, land ownership, animal concentration areas, and land uses and management information for the past several years.

WATERSHED MONITORING

Flow monitoring devices were installed to isolate three watersheds - the South Watershed of about 195 hectares, the East Watershed of 335 hectares and the Manthei Watershed of 22.5 hectares (Figure 1). The larger two watersheds

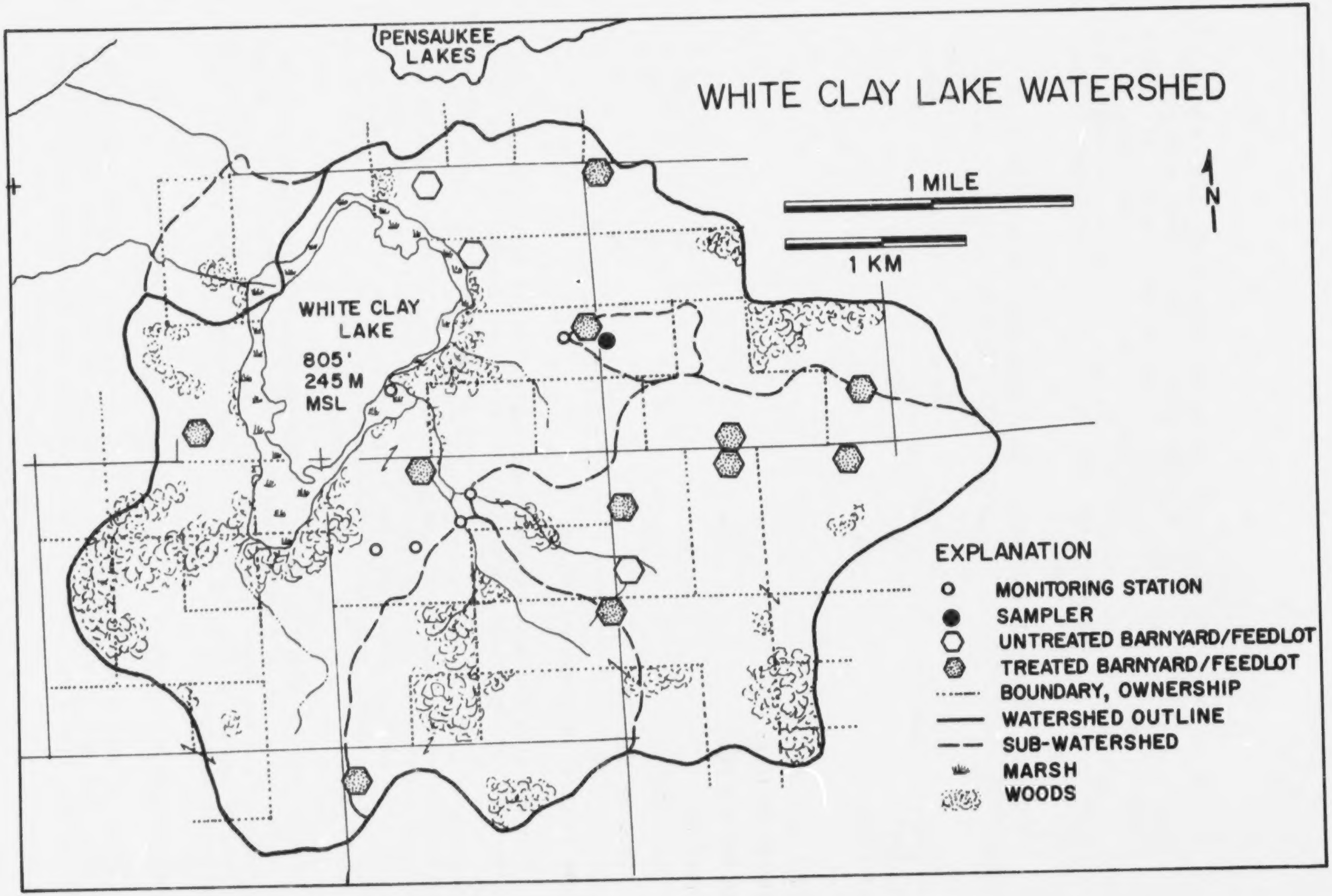
WHITE CLAY LAKE WATERSHED
PENSAUKEE LAKES
WHITE CLAY LAKE
805'
245 M
MSL
1 MILE
1 KM
N
EXPLANATION
○ MONITORING STATION
● SAMPLER
UNTREATED BARNYARD/FEEDLOT
TREATED BARNYARD/FEEDLOT
BOUNDARY, OWNERSHIP
WATERSHED OUTLINE
SUB-WATERSHED
MARSH
WOODS

Figure 1. White Clav Lake watershed map.

were selected to be representative of the soils, topography, and land use of the rest of the watershed as well as other areas of northeastern Wisconsin. A monitoring station on the lake's outlet stream measures output of surface water from the entire watershed. Water samples taken weekly and during runoff events at each station are analyzed for residue, phosphorus, nitrogen, and chloride content. A summary of land uses in each of the watersheds is shown in Table 1.

TABLE 1. SUMMARY OF LAND USES - WHITE CLAY LAKE BASIN (1974-75)

	Entire Basin	South Basin	East Basin	Manthei Basin
Area (ha)	1215	195	335	22.5
% of total	100	16	27.5	1.8
Wooded (%)	23	20	14	0
Littoral Wetlands (%)	6.7[a]	--	--	--
Lake Surface (%)	7.8	--	--	--
Cropped (%)	66	80	85	100
Corn (%)	--	35	25	95
Oats and Hay (%)	--	45	60	5

[a] Some littoral wetlands are wooded.

A survey of ground water movement and quality in the basin (Tolman, 1975) complemented the hydrologic and nutrient transport studies for the lake. Observations on a network of wells and seepage collectors were used to estimate rates of water movement into the lake. Water level recorders showed the relationship between lake level and water table fluctuations. Samples from observation wells were analyzed for chloride, nitrogen and phosphorus content. Samples from private water supplies were analyzed to determine the water quality of the deeper aquifer. Ground water monitoring is continuing on a quarterly basis.

Project weather stations within the watershed provide continuous measurement of precipitation, temperature and relative humidity. Maximum and minimum temperature readings were recorded weekly. Frost depth is monitored using fluorescein tubes (Harris, 1970) at several places in the watershed from December through April.

Watershed Material Transport

Water volume input to White Clay Lake serves as a base for determining nutrient input and hydraulic residence time for the lake. Table 2 shows water contributions from direct precipitation, surface water flow and direct ground

TABLE 2. NITROGEN AND PHOSPHORUS TRANSPORT TOWARD WHITE CLAY LAKE. SUMMARY, 1974.

	Water Volume[a]	%[b]	Mean Concentration mg/l	Total kg	Percent of Total Nutrient
NITROGEN					
Direct Precipitation	0.542	25	1.0	542	10
Metered Surface Water	0.702	35	4.25	2,983	57
Unmetered Surface Water	0.086		4.25	366	7
Ground Water	0.897	40	1.50	1,346	26
	2.227	100%		5,237	100
PHOSPHORUS					
Direct Precipitation	0.542	25	0.25	136	22
Metered Surface Water	0.702	35	0.45	316	51
Unmetered Surface Water	0.086		0.45	39	6
Ground Water	0.897	40	0.15	134	21
	2.227	100%		625	100

[a] in millions of cubic meters input to lake.

[b] percent contribution to total water input to lake, no net change in lake storage during period.

flow for a one-year period, as well as related total nitrogen and phosphorus inputs.

Comparing the relative magnitude of nitrogen compound sources (Table 2) to water sources show that direct precipitation supplied about 10% of the total N in 25% of the water input, surface water supplied 64% of the N in 35% of the water and ground water supplied 26% of the N in 40% of the water.

Table 2 shows estimated total phosphorus inputs to the lake. With 35% of the water input via surface flows came 57% of the total phosphorus. The contributions from direct precipitation are based on only six samples taken in the first 6 months of the year and thereby represent a rough estimate.

Annual totals of water, phosphorus, nitrogen and total residue transport from the East and South Watersheds for 1974-1977 are summarized in Table 3. The water transport in 1977 was the lowest of the 4 years of observations. The outlet stream and the South branch both dried up during July-November of 1977.

Annual residue losses in the two watersheds range from about 45 kg/ha to 750 kg/ha for the 4 years with a peak during the first year which is attributed to site disturbances during construction of monitoring stations. This range of residue transport is considered to be quite low for agricultural watersheds.

Phosphorus areal outputs (below) show considerable yearly range (kg P/ha):

	1974	1975	1976	1977
East	0.64	0.50	2.1	0.25
South	0.56	0.83	0.37	0.01

While these outputs fit within the range of agricultural land outputs listed by Uttormark et al. (1974), the rate for the East Watershed in 1976 appears to be a significant change from past years. Of particular interest is that there was no large increase in output rate from the South Watershed, nor were there similar increases in nitrogen losses. Further analysis of individual runoff events and land management records may help to explain the differences.

The low areal output of materials from both basins in 1977 is related to decreased runoff during this dry year (see Table 3).

Material losses from the Manthei Watershed are summarized in Table 4. The purpose of monitoring this watershed was to estimate material losses from a dairy barnyard. The lower watershed includes the entire basin, terminating at the flow monitoring station. The sampling station of the upper site monitors runoff from about 18 of the 22.5 total ha. The difference in areas includes a dairy heifer operation.

TABLE 3. SUMMARY OF WATER, N, P AND RESIDUE TRANSPORT

	1974	1975	1976	1977
EAST WATERSHED				
Water Volume - cu m	405,000	616,000	682,000	138,825
Residue - total kg	227,400	184,000	197,300	82,420
mean (range) - mg/l	563(10-6210)	299(60-164)	289(29-827)	594(287-1100)
Phosphorus - total kg	214.6	166.9	703	85.3
mean (range) - mg/l	0.53(0.02-6.1)	0.27(0.01-3.47)	1.03(0.01-7.85)	0.625(0.01-3.65)
Nitrogen - Organic kg	624	974	1190	319
mean (range) - mg/l	1.54(0.01-14.9)	1.58(<0.01-4.86)	1.74(0.01-5.58)	
Nitrogen - total kg	1660	2221	2552	785
mean (range) - mg/l	4.10(1.8-15.5)	3.61(1.55-26.8)	3.74(1.57-8.18)	5.65(1.51-16.3)
SOUTH WATERSHED				
Water Volume - cu m	292,000	359,000	194,000	20,438
Residue - total kg	145,400	125,700	78,940	8,845
mean (range) - mg/l	498(10-6000)	350(50-4100)	408(259-727)	433(201-755)
Phosphorus - total kg	109	161.6	72.1	4.05
mean (range) - mg/l	0.36(0.05-7.0)	0.45(0.01-2.93)	0.37(0.01-6.42)	0.199(.01-4.15)
Nitrogen - Organic kg	385	1014	348	25
mean (range) - mg/l	1.32(<0.01-4.6)	2.82(<0.01-15.7)	1.80(0.05-6.93)	1.22(.16-5.95)
Nitrogen - total kg	1323	1989	950	86.2
mean (range) - mg/l	4.53(1.4-8.2)	5.53(0.55-18.8)	4.91(1.32-12.5)	4.23(.97-9.04)

TABLE 4. SUMMARY OF N, P AND RESIDUE TRANSFER - MANTHEI WATERSHED

	1974[a]	1975	1976	1977
UPPER MANTHEI WATERSHED				
Water Volume[b]	---	---	---	---
Residue - total kg	560	9,390	21,090	0.43[c]
mean (range) - mg/l	440(140-2200)	181(10-630)	269(78-712)	131(108-154)
Phosphorus - total kg	0.36	21.7	11.8	0.001[c]
mean (range) - mg/l	0.283(0.16-2.2)	0.418(0.01-1.85)	0.149(0.01-8.20)	0.556(0.30-0.75)
Nitrogen - organic kg				
mean (range) - mg/l				
Nitrogen - total kg	6.39	163	213	0.016[c]
mean (range) - mg/l	5.02(1.8-12.4)	3.14(1.33-8.35)	2.17(0.77-6.67)	5.01(4.01-6.12)
LOWER MANTHEI WATERSHED				
Water Volume	1,274	51,910	78,360	722
Residue - total kg	3,121	11,790	29,210	572
mean (range) - mg/l	2450(160-34,000)	227(100-990)	373(116-937)	792(66-1980)
Phosphorus - total kg	3.01	28.2	140	0.78
mean (range) - mg/l	2.36(0.36-25)	0.541(0.15-1.45)	1.78(0.01-7.15)	1.09(0.05-3.86)
Nitrogen - organic kg				
mean (range) - mg/l				
Nitrogen - total kg	13.5	504	730	14.8
mean (range) - mg/l	10.6(5.9-7.9)	4.40(1.83-9.80)	9.32(1.58-19.3)	20.5(2.33-44.7)

[a] Records from 4/2/74 - 4/12/74.

[b] Volume from Upper is taken as the same as measured at Lower station; consequently mass data from the Upper Station is over-estimated. The area of the upper portion of the basin includes only 18 of the total 22.5 ha area.

[c] Only two samples from Upper Station (versus 54 from Lower Station) representing only 5% of water flow.

Year to year variations in material output are greater than for the larger drainage basins, but the influence of the barnyard area on ambient water quality is readily apparent.

A marsh study including material transport in and out of the littoral zone marsh where the main stream enters the lake will be conducted during 1978-1979.

LAKE STUDIES

White Clay Lake is 95 hectare, dimictic, marl-forming lake with 13 m maximum depth. The lake is underlain by thick glacial drift which fills a deep preglacial valley formed at the contact between Pre-Cambrian igneous and metamorphic rocks and sedimentary rocks of early Paleozoic Age.

The lake exhibits depletion of dissolved oxygen in lower hypolimnetic waters during summer and winter stratification periods. There have not been any fish kills recorded, the plant nuisances are considered to be minimal.

Lake sampling is done cooperatively with the Wisconsin Department of Natural Resources. Monitoring includes monthly measurements of dissolved oxygen and temperature profiles, productivity, algal composition, secchi depth as well as laboratory analysis of water samples from the inlet, lake surface, and 6 meter and 12 meter depths within the lake. Analyses are made for chlorophyll *a*, nitrate, nitrite, ammonium-N, organic nitrogen, reactive phosphorus, total phosphorus, calcium, magnesium, sodium, potassium, chloride, sulfate, alkalinity and pH. Table 5 summarizes water analyses for 1973-1977. Selected dissolved oxygen, temperature and chlorophyll *a* isopleths are shown in Figures 2 through 4.

The overall nitrogen loading directed to the lake in 1974 was 5.5 g N per m^2. Direct precipitation and ground water supplied 1.99 g/m^2 alone. Vollenweider (1968) suggested an annual loading of 2.0 g/m^2 for a lake of this depth as "dangerous" levels of reactive or available nitrogen. Even if only 60% of the total N were biologically available, the loading to the lake would be above Vollenweider's criterion.

Phosphorus loading directed to the lake was also estimated. Phosphorus is the key element in lake eutrophication considerations. It may stimulate nuisance aquatic plant growth, but it is also an element that may be controlled.

Converting the total phosphorus transport to a lake loading yields 0.66 g/m^2. Surface water inputs alone supply 0.38 g/m^2. Comparison of these numbers with Vollenweider's (1968) dangerous levels (0.13) suggests that the lake is under stress. Assuming that 30% to 70% of the total P is biologically reactive, the apparent loading is still excessive. It must be noted that an assessment of P transport across the marsh fringe surrounding the lake has not been made so that the lake-P loading estimates may be somewhat high.

TABLE 5. SUMMARY OF CHEMICAL ANALYSES - WHITE CLAY LAKE

Sample Identification[a]		1973		1974		1975	
		1 (10)		9 (1,2,3,5,6,7,9,10,11)		12 (1,2,2,3,5,6,7,7,10,10,11,12)	
Component[b]	Lake Sample Depth, m	Inlet	Deep Hole	Inlet	Deep Hole	Inlet	Deep Hole
Secchi depth (m)	----	---	2.4	-----	2.32(0.7-3.6)[c]	-----	2.96(2.1-3.8)
Organic N	0.5	1.3	1.2	0.97(0.37-3.4)	0.88(0.64-1.1)	0.59(0.19-2.0)	0.70(0.44-0.91)
	6	---	---	-----	0.84(0.65-1.08)	-----	0.68(0.32-1.2)
	12	---	1.2	-----	0.84(0.53-1.36)	-----	0.73(0.29-1.3)
Total N	0.5	3.4	1.3	2.9(2.2-6.1)	1.3(0.8-2.8)	4.0(1.1-7.6)	1.6(0.6-6.5)
	6	---	---	-----	1.4(1.1-1.8)	-----	1.1(0.8-1.8)
	12	---	1.31	-----	1.8(1.2-3.0)	-----	1.3(0.8-2.2)
Reactive P	0.5	0.16	0.05	(0.006-0.3)			
	6	---	---	-----		-----	
	12	---	0.04	-----		-----	
Total P	0.5	0.27	0.19	0.16(0.02-0.65)	0.07(0.02-0.13)	0.10(0.01-0.63)	0.04(0.01-0.13)
	6	---	---	-----	0.05(0.02-0.07)	-----	0.03(0.01-0.07)
	12	---	0.14	-----	0.07(0.03-0.11)	-----	0.05(0.02-0.14)
Specific conductance[d]	0.5	557	347	447(353-738)	374(260-428)	566(373-895)	378(326-450)
Ca	0.5	70	40	55(27-93)	43(31-59)	46(22-80)	38(16-87)
Mg	0.5	36	27	33(19-41)	28(19-40)	38(16-50)	31(24-50)
Na	0.5	10	6	6(3-13)	5(3-6)	6(1-12)	4(1-8)
K	0.5	8	6	6(3.1-9.3)	6(4.4-7.5)	4(1.7-11.2)	5(1.7-7.1)
Sulfate	0.5	28	11	30(23-35)	18(15-20)	27(25-32)	18(13-27)
Cl	0.5	23	12	21(14-26)	13(10-15)	26(15-26)	13(11-15)
pH units	0.5	8	8	- (7.6-8.3)	- (7.6-8.3)	- (7.6-8.9)	- (7.6-8.5)
Alkalinity[e]	0.5	250	170	247(95-344)	194(114-400)	236(156-317)	167(90-202)

[a] Total number of samples and months during which sampling took place.

[b] mg/l except as noted.

[c] Arithmetic mean and range.

[d] micromhos/cm @ 25°C.

[e] mg $CaCO_3$/l.

TABLE 5. (continued)

Sample Identification[a]		1976		1977	
		8(2,3,4,6,7,7,8,12)		19(1,2,4,4,5,5,5,6,6,7,7,8,8,9,9,10,10,11,12)	
Component[b]	Lake Sample Depth, m	Inlet	Deep Hole	Inlet	Deep Hole
Secchi depth (m)	---	-----	3.03(1.2-5.4)[c]		
Organic N	0.5	0.68(0.20-1.01)	0.65(0.52-1.06)	0.52(0.20-1.04)	0.78(0.43-1.13)
	6	-----	0.70(0.61-0.86)	-----	0.80(0.35-1.22)
	12	-----	0.74(0.40-0.99)	-----	0.94(0.61-1.42)
Total N	0.5	3.23(1.56-5.4)	0.97(0.60-1.29)	3.38(1.97-6.26)	0.94(0.40-1.21)
	6	-----	0.87(0.68-1.2)	-----	0.95(0.43-1.35)
	12	-----	1.68(0.95-2.78)	-----	1.90(0.80-4.29)
Reactive P	0.5	-----	---------	0.03(0.005-0.059)	0.016(0.004-0.056)
	6	-----	---------		0.012(0.004-0.041)
	12	-----	---------		0.017(0.004-0.046)
Total P	0.5	0.68(0.01-0.19)	0.04(0.02-0.11)	0.07(0.02-0.19)	0.034(<0.01-0.07)
	6	-----	0.05(0.03-0.13)	-----	0.032(0.01-0.07)
	12	-----	0.05(0.02-0.16)	-----	0.050(0.01-0.10)
Specific conductance[d]	0.5	567(419-673)	364(331-404)	573(531-619)	417(331-501)
Ca	0.5	57(34-75)	35(23-47)	56(46-68)	38(27-42)
Mg	0.5	42(36-46)	30(26-33)	42(31-46)	31(21-37)
Na	0.5	5.3(1-19)	4.7(1-18)	4.0(1-7)	4.2(1-8)
K	0.5	3.8(1.7-8.6)	5.4(2.7-14)	4.3(1.8-6.7)	4.6(3.5-5.1)
Sulfate	0.5	31(25-34)	19(17-23)		
Cl	0.5	24(19-33)	12(3-15)		
pH units	0.5	- (7.4-8.1)	- (7.7-8.1)	- (7.4-8.4)	- (7.5-8.7)
Alkalinity[e]	0.5	225(120-309)	160(137-188)	239(198-288)	179(137-242)

[a] Total number of samples and months during which sampling took place.

[b] mg/l except as noted.

[c] Arithmetic mean and range.

[d] micromhos/cm @ 25°C.

[e] mg $CaCO_3$/l.

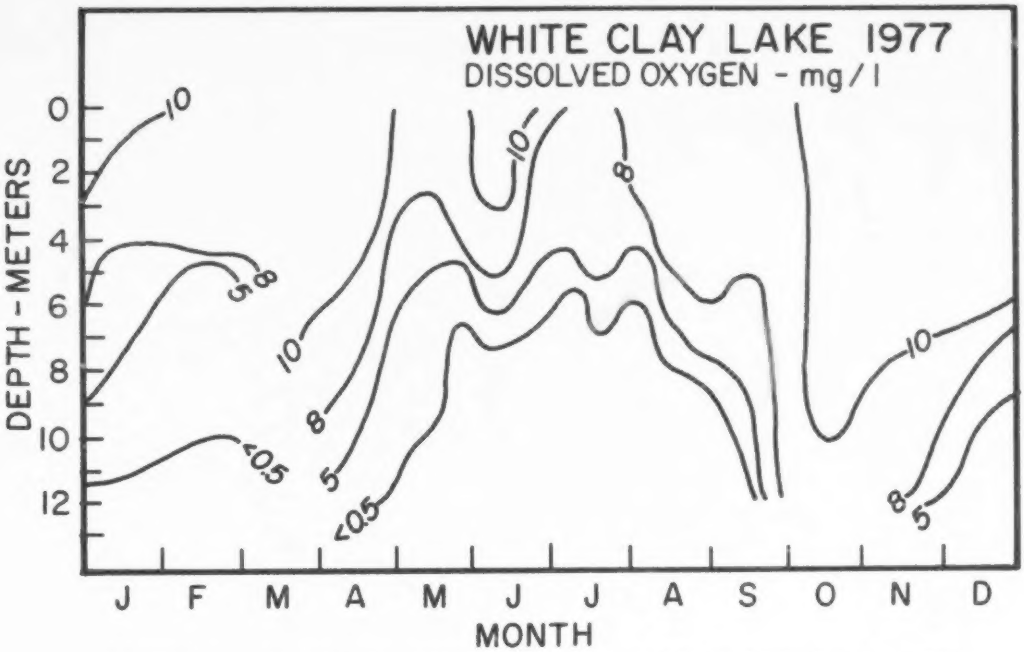

Figure 2. Dissolved oxygen isopleths—White Clay Lake, 1977.

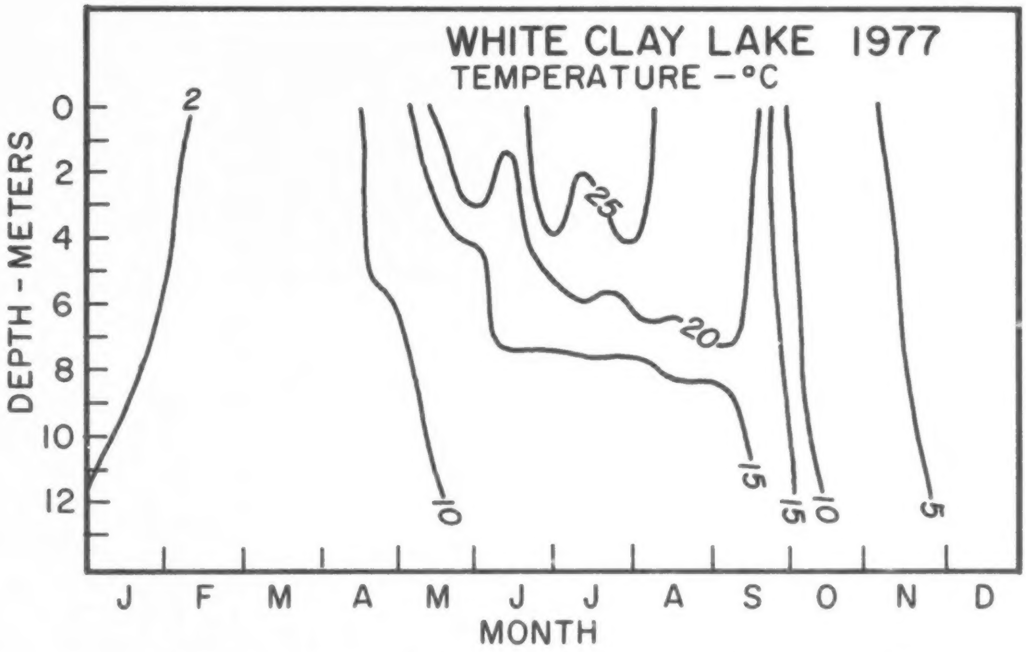

Figure 3. Temperature isopleths—White Clay Lake, 1977.

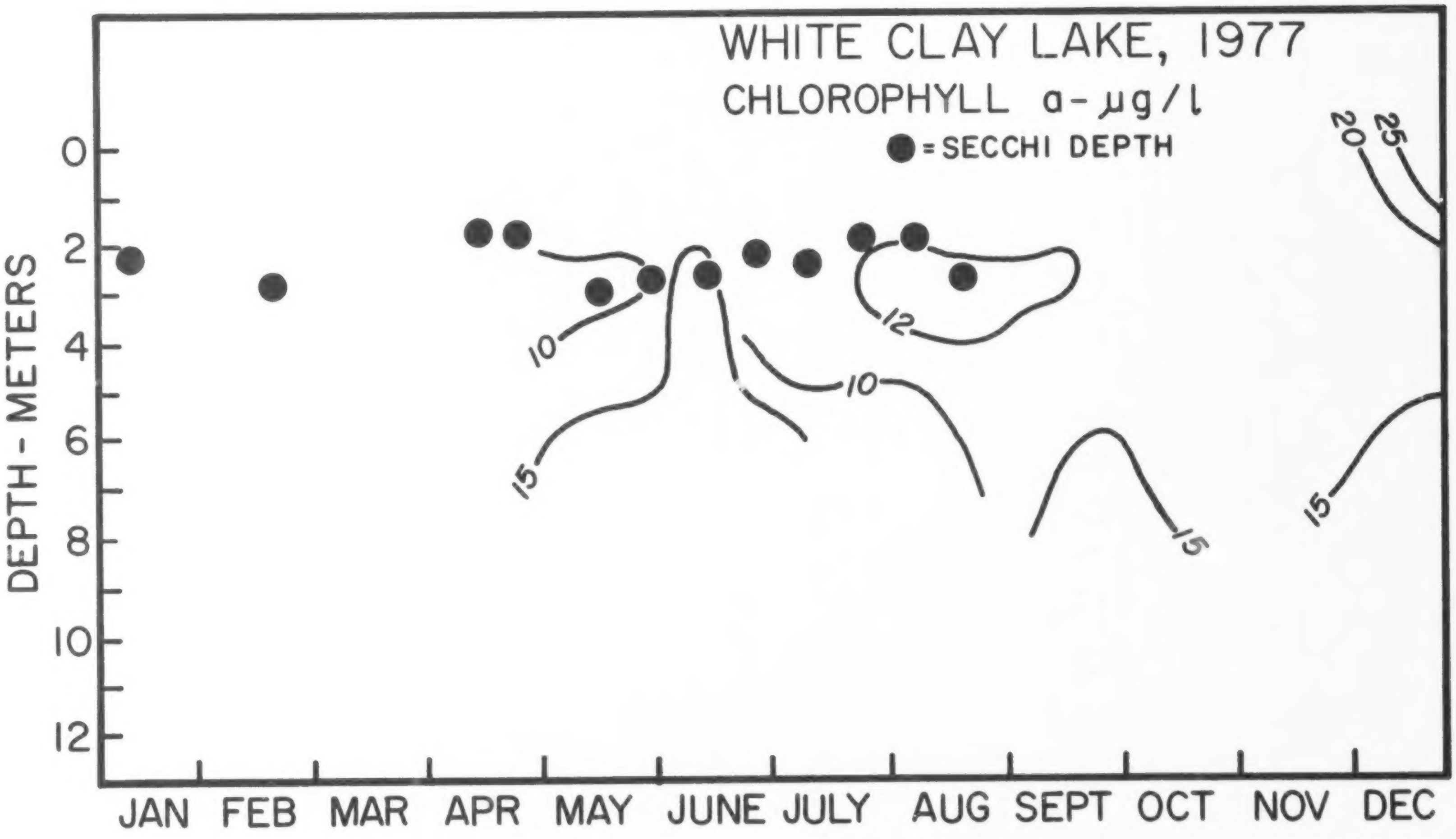

Figure 4. Chlorophyll and Secchi Depth—White Clay Lake, 1977.

OTHER STUDIES

Cooperative research between the University of Wisconsin-Madison and the USDA Sedimentation Laboratory (Bubenzer et al., 1974) was initiated to investigate erosion and deposition processes on the White Clay Lake Watershed using Cesium-137 as the tracer. Preliminary results indicate an overall erosion from the cultivated areas with some deposition on the upland watershed. Much of the deposition from the watershed appears to be taking place in the marsh fringe around the lake. Significant Cesium-137 concentrations have been found at the 50 centimeter depth within the marsh while depths of 10 centimeters or less have been observed in the adjacent littoral zone of the lake. The results indicate that Cesium-137 can be used as a "tag" to measure both the erosion and deposition of sediments in agricultural watersheds such as White Clay Lake.

SUMMARY

The development and implementation of the White Clay Lake Management Plan is an example of effective cooperation between individual citizens, local units of government, state and federal agencies, and the University System. The effort is providing valuable insights into many of the questions being raised about the implementation of rural nonpoint source pollution control programs.

By its very nature, nonpoint source pollution is a problem which requires the interaction of a variety of agencies. The number of agencies involved results from the historic separation between those which deal with land resource problems and those which deal with water resource problems. Partnership between these diverse interests is critical if water quality problems are to be solved through the implementation of land management plans.

It would appear now that responsibility for the implementation of rural nonpoint source pollution programs will be vested in the traditional federal agencies, namely, the SCS and ASCS, working with local SWCDs. Cost sharing money will be available both for nonpoint control measures or Best Management Practices for traditional conservation measures. At White Clay Lake the Shawano County SWCD has been a co-sponsor of the project since its inception, although the Lake Protection District has, since its formation, served as the focal point for identifying nonpoint source problems and for allocating funds for improvements designed to solve those problems.

From the experience at White Clay Lake, it would appear that this mechanism—the creation of a Lake Protection District—can be an effective means of dealing with critical nonpoint source areas. In watersheds of reasonable size, it affords local residents the opportunity to develop and implement land management plans designed to improve water quality. Of further importance is that Lake Protection Districts have the power to tax. The White Clay Lake District has never levied a tax, but the authority is there and it might be a way to raise money to supplement funds available from other sources for nonpoint source pollution control.

When Lake Protection money became available, members of the White Clay Lake District agreed to use it for their most critical nonpoint problems—barnyards and feedlots—and to use monies from the ACP program for cost sharing the installation of conservation practices on croplands. The reasons for this were eminently practical—there was not enough Lake Protection money to do everything so investments of these funds were directed toward the most critical problem areas. Additionally, barnyard work is expensive and in all cases the money required far exceeded the traditional $2,500 per farm per year limitation of the ASCS program. This innovative approach might well be applied to the allocation of newly authorized federal nonpoint source control money.

Designers of nonpoint source pollution control programs are currently debating the question of mandatory vs. voluntary control programs. At White Clay Lake the District was able to share 90% of the cost of control structures, a figure somewhat higher than that envisioned for new nonpoint programs. Of the farms with livestock in the watershed, all but 3 were improved using project funds. This is a cooperation rate of about 83%. It should be noted, however, that two of the noncooperating farms are located directly on the shore of the lake and that both have large livestock operations for which adequate protection against sediment and nutrient movement is not provided.

The University has played an important role in the White Clay Lake effort since its inception. Responding to concern expressed by residents of the watershed about the water quality of the lake, University personnel helped hustle grants, design and install the monitoring network to quantify movement of sediment and nutrients from agricultural operations toward the lake. Data from this work served to meet the feasibility requirements of Chapter 33, thus making the Lake Protection District eligible to apply for funds to implement a management program. Research work showed that even though the water quality of the lake itself was good, nutrients were moving to the lake in amount well in excess of those considered to be safe for maintaining present lake quality. Attention was focused on land activities, as major changes in the in-lake system were not expected during the course of the study. Excessive nutrient loadings were the basis for the protection program rather than changes in water quality.

Project activities are continuing. Now that protective measures have been installed in barnyards and feedlots, on streambanks and on cropped lands, monitoring is being continued to assess the effectiveness of these practices. The marsh area, through which much of the water going into the lake moves, is being studied to determine its effectiveness as a nutrient and sediment trap. Long-term surveillance is essential to determine change in lake water quality and reductions in pollutant loadings.

The White Clay Lake experience has been valuable in many ways. It is a good research tool providing insights into environmental problems resulting from agricultural operations and the movement of sediments and nutrients into lakes and streams. It is an excellent educational tool not only for the residents of the watershed but also for the many students, elected officials and citizens who have toured the project area. It is a fine demonstration of local people working with a number of agencies and institutions to solve specific problems.

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SOCIO-ECONOMIC IMPACT OF LAKE IMPROVEMENT PROJECTS AT MIRROR/SHADOW LAKES AND WHITE CLAY LAKE

by

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HISTORY OF LAKE USE

Mirror and Shadow Lakes are small natural lakes located in the City of Waupaca (1970 pop. = 4,342). The city is the county seat and largest city in Waupaca county (1970 pop. = 37,780). Principal economic activities in the county are agriculture and tourism with some light manufacturing. Mirror Lake is 13 acres and Shadow Lake is 40 acres in size. At one time ice was harvested from Mirror Lake but the practice was stopped in the 1950's when the ice began to smell (decaying algae) as it melted. Both lakes have been used for over 100 years for water recreation. Water is supplied to the lakes by groundwater and until recently by storm sewers.

White Clay Lake is a natural lake (≅250 acres) in Shawano County (1970 pop. = 6,488). The county seat is Shawano (1970 pop. = 6,488), located about nine miles west of the lake. Green Bay (1970 pop. = 87,809) is less than one hour driving time to the southeast. The village of Cecil (1970 pop. = 369) is located two miles northwest. The lake has been used for marl production and recreation. The main economic activities of the county are agriculture and tourism. White Clay Lake is located in a small watershed of 3000 acres that is used almost exclusively for agriculture. There are ~30 landowners in the watershed and 14 own livestock. There is one small resort and a few other non-farm residences. Most pollutants entering the lake probably originate from farmlands.

LAKE USERS

Mirror and Shadow Lakes are most heavily used for swimming in summer by local residents especially mothers with children. The Shadow Lake beach is the only beach in the city. There is regular but limited fishing and non-power boating activity. The lakes are also used for ice skating in the winter. Some day-users from surrounding communities are attracted to the lakes for picnicking and water sports but overnight tourists are not common. The Waupaca Chain of Lakes is located a few miles southwest but its use is dominated by power boating and fishing activities of second home owners and tourists. Shadow Lake is part of a large city park which serves as a focal point

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of community recreation especially in summer. In July and August of 1977 the average mid-afternoon count of swimmers was 77.2. On July 14 at 2:00 p.m. 350 swimmers crowded on the beach. The maximum number of boaters and fishermen on the lake at any one time during the day averaged 2.6 boaters and 4.6 fishermen. It should be noted that these figures account for the number of recreationists at the single time of day when use was highest. Total users per day may be two-three times the figures given. There did not appear to be any increase in activity during the weekend. More refined data on usage will be collected in 1978-79.

White Clay Lake is used for fishing year around. There is some pleasure boating in summer (speed of boats is controlled by town ordinance) and hunting in fall. The users appear to be mostly local residents and day-users from surrounding cities. Shawano Lake, a major tourist attraction and second home center a few miles west of White Clay Lake, attracts most of the powerboating and waterskiing. White Clay Lake appears to be viewed as a quiet complement to the noise and bustle and surface water user conflicts of Shawano Lake. When morning and evening observations are combined, and average of 5.2 boats per day were being used for fishing from August through November of 1977.

In December and January an average of 17.6 people were ice fishing each day. A major local event is the ice fishing derby which attracted 155 ice fisherpersons on a Sunday in January. With this exception, the amount of weekend activity is not particularly pronounced compared to weekday activity. It should again be noted that all recreationists were not counted. Mid-day recreationists (particularily in summer) were not observed. However most fisherpersons were probably noted since fishing is concentrated in the morning and evening. More refined data on usage will be collected in 1978-79 from the beginning of spring fishing to the end of the ice fishing season.

LAKE MANAGEMENT DISTRICTS

Mirror and Shadow Lakes were part of the Inland Lake Demonstration Project of the Wisconsin Department of Natural Resources (DNR) and University of Wisconsin Extension. The study of these lakes, funded by the Upper Great Lakes Regional Commission, revealed that storm sewers were the primary source of the nutrients (phosphate) that were feeding increasing growths of weeds and algae. In 1974 the Wisconsin Legislature enacted Chapter 33 of the Wisconsin Statutes which enabled local communities to form a special purpose unit of government to manage their lake(s). The Waupaca City Council created one of the first lake management districts.

The City Councilmen also serve as the commissioners of the district. In 1975 they voted to undertake a restoration project which consisted of storm sewer diversion to prevent the entry of new nutrients, alum treatment to inactivate nutrients already in the lakes, and aeration of Mirror Lake to prevent fish-kills. They applied and received $130,000 in state funds. Through the DNR they also applied for EPA funds under Section 314 (Clean Lakes Act) of Public Law 92-500. In the first set of awards under Section 314, Waupaca was awarded a grant of $215,000 in January 1976. Additional local matching funds were necessary in the amount of $80,000. The district electors had voted a tax levy of 0.9 mils on the taxable property (equalized value =

$41,000,000) for two consecutive years. The city also decided to spend extra money to repave the streets where new storm sewer lines were constructed. Because the district had voted the tax in 1975, conditional on an EPA grant, work began almost immediately after the EPA announcement and storm sewer construction was completed in 1976.

White Clay Lake was also the subject of previous study involving the Wisconsin DNR, University of Wisconsin Extension, Upper Great Lakes Regional Commission, Soil Conservation Service, U.S. Geological Survey, U.S. Agricultural Research Service, and the Shawano County Soil and Water Conservation District. The lake is still of high quality but the study indicated a high potential for degradation from agricultural runoff if farming practices were not changed. With encouragement from Extension specialists, the farmers in the watershed asked their town board to create a lake management district under Chapter 33 for the purpose of protecting White Clay Lake.

All the land in the watershed was included in the district formed in late 1974. In 1975 the district received a state grant and through DNR applied for an EPA grant. White Clay Lake was also among the first set of awardees under the federal Clean Lakes Act in January 1976. EPA contributed $107,000, DNR contributed about $100,000, and the district contributed in-kind services to complete the matching requirements. The Shawano County Agricultural Stabilization and Conservation Service has provided the accounting service.

A number of barnyards were resloped and manure storage facilities built in the fall of 1976. Most of the other farmers asked for similar construction work in 1977. During this process Tom and Dave Brunner, young and progressive farmers, provided leadership within the community. Their farm became a local and statewide model of land and manure management practices. The lake district petitioned the town board for self governance under 1976 amendments to Chapter 33 of the Wisconsin Statutes. In the ensuing election Tom Brunner was elected chairman of the district. Subsequently, he was also elected to the town board.

PHILOSOPHY OF ANALYSIS

An investigation of the socioeconomic impacts associated with lake rehabilitation/protection requires a very broad conception of the stimulus producing these impacts. The investigation cannot be limited to the impact of the actual physical intervention of the technology; the investigation must view the project as a social process which began when local citizens began to see problems with their lake, organized to combat these problems, took action, and are now "reaping the benefits" of their investment and the investment of funding agencies. The process actually continues on into the future. Under ideal conditions data would be gathered at several points in time. Baseline data would be gathered before the prospect of a lake project had begun to "contaminate" perceptions. Impact data would be gathered during the project, immediately after the project, and several years later when limnological changes had fully manifested themselves. While it would still be difficult to separate out other causal agents, a comparison of baseline data and impact data would be the best basis for evaluation. For obvious reasons this investigation is limited to one point in time--immediately after the physical intervention. This requires comparisons with other control groups and the use of models to quantify impacts.

SPECIFIC RESEARCH QUESTIONS

Economic evaluation questions relate to the benefits and costs associated with efforts to reduce and correct lake pollution; in these cases, overfertilization is the problem. The following four questions are being addressed:

1. What are the recreational benefits associated with an increase in water quality?
2. What are the benefits accruing to affected property owners because of increases in water quality?
3. What are the aesthetic impacts and how are the trade-offs between these and economic benefits viewed by the public?
4. What are the costs to the agricultural sector for compliance with alternative preventive and/or remedial actions to pollution?

Sociological evaluation questions are more process oriented than product oriented. The specific questions span a variety of quantative and qualitative parameters and the impacts span a period of years. The following is an attempt to categorize the questions and structure the data to the degree possible:

1. What are the necessary institutional conditions for undertaking a lake restoration project?
    a. What involvement is necessary by "the general public", local property owners, and local officials?
    b. What legal powers are necessary to raise revenue for local matching of federal grants?
    c. What types of local leaderships are necessary and how does such leadership develop?
    d. What types and degrees of support from the media and educational institutions are necessary?
    e. What is the optimal institutional arrangement and division of responsibility/authority between federal, state, and local "partners" in a lake restoration effort?
2. Who is being impacted and what is the differential impact on various segments of the population?
    a. What is the perception of the changes in water quality and in distribution of project benefits and costs?

b. Who cares about water quality and what aspects do they care about?

c. Who uses the water and who owns the shoreline?

d. Who has the option of substitution and at what cost?

e. How have patterns of interaction changed or been maintained between neighbors, kin, and recreating groups?

3. What is the long term impact on ecological awareness and participatory democracy?

   a. Do local residents better understand ecological principles and lake systems?

   b. Have attitudes toward state and federal agencies improved or deteriorated?

   c. Has the stimulus of the project developed a sense of control of community destiny and personal efficacy or contributed to the fatalism of "small town in mass society"?

   d. Has community cohesion suffered or increased as a result of the project?

4. Would the residents and the local leaders do it over again if they made the decision now?

THEORETICAL APPROACH

Economic analysis will be guided by four models which correspond to the four economic questions noted earlier:

1. To evaluate recreational benefits a travel-cost model of the Clawson genre is geing employed, but with observations based on individual observations rather than grouped data. This will allow for the inclusion of variables, such as cost and distance, that normally cause multicollinearity problems. The general form of the model employed to represent the demand relationship is:

$$V_{ij} = \alpha_j + \sum_{k=1}^{n} \beta_{ij} x_{ij} + e_{ij}$$

   where V_{ij} is the number of visits by decision-making unit i to lake j, x_{ijk} is the value of the independent variable k for the decision-making unit i on lake j, and e_{ij} is the error term. The primary objective is to produce a statistical demand curve with reliable estimates of the structural parameters--particularly those of the cost variable from which the resource variable is derived, and those of the water quality variable which is used to determine the economic significance of a water quality change.

The methodology employed extends previous efforts by incorporating the recreator's perception of water quality directly into the model. The model also links these perceived subjective ratings to the objective water quality ratings (Lake Condition Index) of limnologists.

2. To estimate those benefits capitalized in property values an existing model as developed by Dornbusch et al. will be applied. This model depicts the benefits of improved water quality as decreasing proportionally with the reciprocal of the distance to the water body. Application of the model requires a water quality expert's statement of both present and predicted levels of water quality expressed in terms of the components used to describe the Perceived Water Quality Index (PWQI). The PWQI of the water quality expert along with information on the water body type and the degree of public access and use, is used to determine the PWQI value which would be perceived by residents at the site. To obtain a value for the coefficient of the distance-to-water term in the expression yielding the percent change expected in prices of properties at the site, the value of the PWQI and water body type are utilized.

   Thus, the change in price expression is:

$$\Delta P\% = b_o + b_1 \, (^1/DW)$$

   where $b_o = -b_1 \, (^1/DW)$

$$b_1 = e^{6.398} \, (PWQI_{res})^{.492} \, e^{1.18 \text{ WBT Lake}} \, e^{.991 \text{ WBT Bay}}$$

   DW = maximum distance from water up to 4,000 feet.

   WBT Lake = dummy variable with value of 1 if water body is a lake and zero otherwise.

   WBT Bay = dummy variable with value of 1 if water body is a bay and zero otherwise.

   The change in price is now applied to zones where the number of homes and average home price are used to calculate total price change.

3. To estimate aesthetic impacts a model is proposed which consists of a hierarchical array of elements, social goals, subgoals, social indicators, and action (or decision) variables. A change in any one element of the model is, in general, related to a change in all

other model elements. An expression which states a relationship between two elements is called a connective. Goals are further and further broken down into their component parts until they are represented by measurable parameters (social indicators). It is these social indicators that are impacted by public action, i.e., a water improvement program. To establish the ultimate impact of a public action on the attainment of a social goal, viz. aesthetics, it is necessary to establish the relative weight of the components comprising a higher level goal or subgoal and to establish the functional form of the connectives between the lower and the next higher step in the hierarchy.

4. To estimate farm level impacts of institutional alternatives designed to modify operator behavior a linear programming model will be employed. This model is based upon existing management practices and technology in order to capture the status quo mix of agricultural activity. The economic model provides the land use configuration necessary for running a hydrologically oriented simulation model which predicts both total storm watershed soil loss and the concentration of sediment in watershed drainage water. Having captured the status quo land use configuration and its attendant sediment yield a set of institutionally determined parameters, such as alternative levels for cost-sharing minimum tillage systems, low interest loans for terraces, technical assistance and education, and tolerable soil loss limits are introduced into the economic model. The economic, administrative, and land use implications of these alternatives can then be examined.

Sociological analysis cannot be defined by a neat set of models or equations. No single theoretical perspective adequately addresses the range of impacts--changes in social structure, values, attitudes, and behavior of the impacted population. The following perspectives are influencing the research design but knowledge of local conditions is also being used to select appropriate parameters:

1. Under the Northwest Ordinance and the Wisconsin Constitution, lakes are held in trust for the public but little legal provision was made for their management. In many ways lakes suffer from the "tragedy of the commons" and theories of managing the commons can be used as a framework to discuss the respective rights and responsibilities of public users, riparians, local officials, and agency bureaucrats. (Managing the Commons, eds. Garrett Hardin and John Bader.) Alternative institutional arrangements between these groups will receive substational attention.

2. Mancur Olson's Logic of Collective Action provides a departure point to analyze the interplay of groups needs (to manage the lake) and individual motivations to "let George do it" unless separate and selective incentives are provided and personal efficacy is demonstrated.

3. The theme of adaption developed by Honey and Hogg may be most useful to large technological interventions but it is also useful to assess the impact of less traumatic lake projects on the way individuals and institutions relate to their natural resource base (cultural-environmental relationship). For example, the willingness of White Clay Lake area farmers to build manure storage facilities may be due to economic incentives, ecological sensitivities, and the threat of non-point pollution abatement regulations. The lake protection project may be perceived by the agricultural community as a way of coping with future disruption.

4. The entire lake restoration process is a special type of community development. This perspective provides a framework to analyze leadership development, consensus building, and public participation.

DATA NEEDS

Statewide survey will be conducted to provide:

1. data for development of the recreational model,
2. data for the aesthetic model, and
3. comparative data for the sociological analysis.

A probability sample will be drawn for telephone interviews with a randomly selected adult in the household. Since not all Wisconsin adults will have recreated in one of Wisconsin's 1100 largest lakes (lakes over 100 acres have been rated for water quality on a scale from 0-23) during the previous year, the initial sample size must be expanded to provide sufficient number of lake recreationists for the recreation model.

Farm operators and other residents of the White Clay Lake watershed will be personally interviewed to:

1. obtain information on the farm operation,
2. ascertain degree of involvement with and attitude toward the lake district/project,
3. obtain data which can be compared to the statewide survey,
4. determine use of the lake.

Waupaca riparian/property value data will require evaluations on the water quality components that comprise the PWQI from the limnologists associated with the Mirror/Shadow Lakes project. Information regarding number of properties within each zone from the water body can be obtained through on-site observations. Information regarding property values can be obtained from real estate offices, tax rolls, or residents themselves. More than one source may be chosen for comparative purposes. Personal interviews will be conducted with the riparians to:

1. obtain information on perception of property values,

2. ascertain degree of involvement with and attitude toward the lake district/project.

3. obtain data which can be compared with the state-wide survey, and

4. determine use of the lake.

Recreationists will be interviewed at both sites to:

1. obtain data which can be compared to the state-wide survey,

2. determine use of the lake, and

3. ascertain patterns of recreational behavior and group interaction.

Ethnographic information has been and will continue to be obtained by the research team through extensive contact with community leaders, as a by-product of the personal interviewing conducted by project personnel, and related case study investigation of documents and media reports.

STATUS

Statewide survey will be conducted in September immediately following the Labor Day close of the summer recreation season. The schedule is in the process of completion at the present time and has been reviewed by Russell Gum and Louise Arthur of the USDA Economic Research Service. Daniel Bromley, Thomas Heberlein, Basil Sharp, and Douglas Yanggen of the University of Wisconsin, and Michael Patton of the University of Minnesota will review this schedule as well as the other schedules noted below.

Farm operators' schedule has been used in another related project in Wisconsin and with some additions is very nearly completed. Interviews with farmers are scheduled for March 1978 before spring planting begins.

Waupaca riparian/property value data will be collected later in 1978.

Waupaca residents will be interviewed later in 1978.

Recreationists will be interviewed over an entire year since activities occur in each season. The schedule will be finalized in April of 1978 and interviewing will begin with the beginning of the spring fish season in May and continue through the ice fishing season next winter.

Ethnographic information has been informally collected by the project director since 1974. A systematic effort will begin in March of 1978 when the research team begins to spend extended periods of time in the community. This type of information will continue to be gathered until the final report is written.

APPLICATION OF RESULTS

The U.S. Congress reaffirmed in Public Law 92-500 that clean water was a national goal. By definition such a goal is considered to contribute to the social well being of our society. It is a desired state of affairs that is sufficiently broad and multifaceted to insure unanimity as to its appropriateness.

However, as the goal became more specific there is less unanimity; with limited resources choices must be made regarding which water to clean up (or keep clean) and to what degree of purity. Should resources be concentrated on the Great Lakes, inland lakes, major rivers, streams, or groundwater? Should point or non-point sources receive greater attention? Is agricultural, industrial, or residential pollution most severe and which is easiest to correct? Should highly eutrophic lakes be rehabilitated or should high quality lakes be protected? Should lakes in residential areas or lakes supporting a hospitality industry receive priority? How important is local commitment and a legal infrastructure?

This research is not intended to answer all the above questions but should help decision-makers at all levels of government answer some of them. It is inappropriate to decide public policy by taking a poll but the information from the statewide survey will show the relationship between recreational activity/satisfaction and lake water quality. It wil also provide information on lake users--their characteristics, knowledge, attitudes, and aesthetic preferences.

The other surveys will provide specific information on the benefits and costs associated with two lake projects in communities where overnight tourists are not a major user group. It will also provide information on changes in knowledge, attitudes toward government and citizen participation, and community leadership. Finally it will provide a list of necessary institutional conditions and recommend intergovernmental interaction for undertaking a lake restoration project.

The results will not provide a single formula which can be applied to several candidate lakes to rank them for funding. In the opinion of the authors it is neither possible nor desirable to abdicate legislative and agency judgement to a mathematical model. It seems appropriate that the local community, state government, and EPA continue to make individual judgements on project viability and cost effectiveness. The results of this research should assist those judgements but not replace them.

EVALUATION OF LAFAYETTE RESERVOIR RESTORATION PROJECT

by

M. W. Lorenzen, F. M. Haydock, T. C. Ginn*

INTRODUCTION

Sections 314/104(h) of the Federal Water Pollution Control Act Amendments (PL 92-500) of 1972 are directed toward nationwide restoration and protection of lake water quality. Under this program, federal grants are awarded to local agencies on a 50:50 matching basis to fund lake restoration projects which qualify. The East Bay Municipal Utility District (EBMUD) was awarded a demonstration grant under this "Clean Lakes Program" and proposed to implement a lake restoration project at Lafayette Reservoir. The proposed project includes hypolimnetic aeration to provide a suitable habitat for cold water sport fish and alum treatment for nutrient inactivation to limit algal growth.

Tetra Tech, Incorporated, will conduct an independent study to evaluate the restoration project. The purpose of this study is 1) to monitor water quality conditions before, during, and after restoration, 2) to analyze these data in conjunction with the application of a water quality ecological model to elucidate the mechanisms of water quality improvement, and 3) to evaluate the technical characteristics of the restoration system for potential application elsewhere.

LAFAYETTE RESERVOIR

The reservoir is located in Lafayette, California, approximately 20 miles east of San Francisco (Figure 1). Lafayette Reservoir and its watershed are owned and operated by the EBMUD as a recreational facility and emergency standby water supply. It was created in 1929 when an 92-inch earth filled dam was built. Since it is situated close to the Bay Area Rapid Transit (BART) Station and Interstate Highway 24, it is readily accessible to San Francisco Bay Area Residents.

Weather conditions in the area are generally mild. Annual precipitation averages 26 inches. The topography of the watershed is shown in Figure 2. The drainage basin encompasses only 1.3 square miles (830 acres) most of which is undeveloped park and recreational area.

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* Tetra Tech, Inc., Lafayette, California 94549

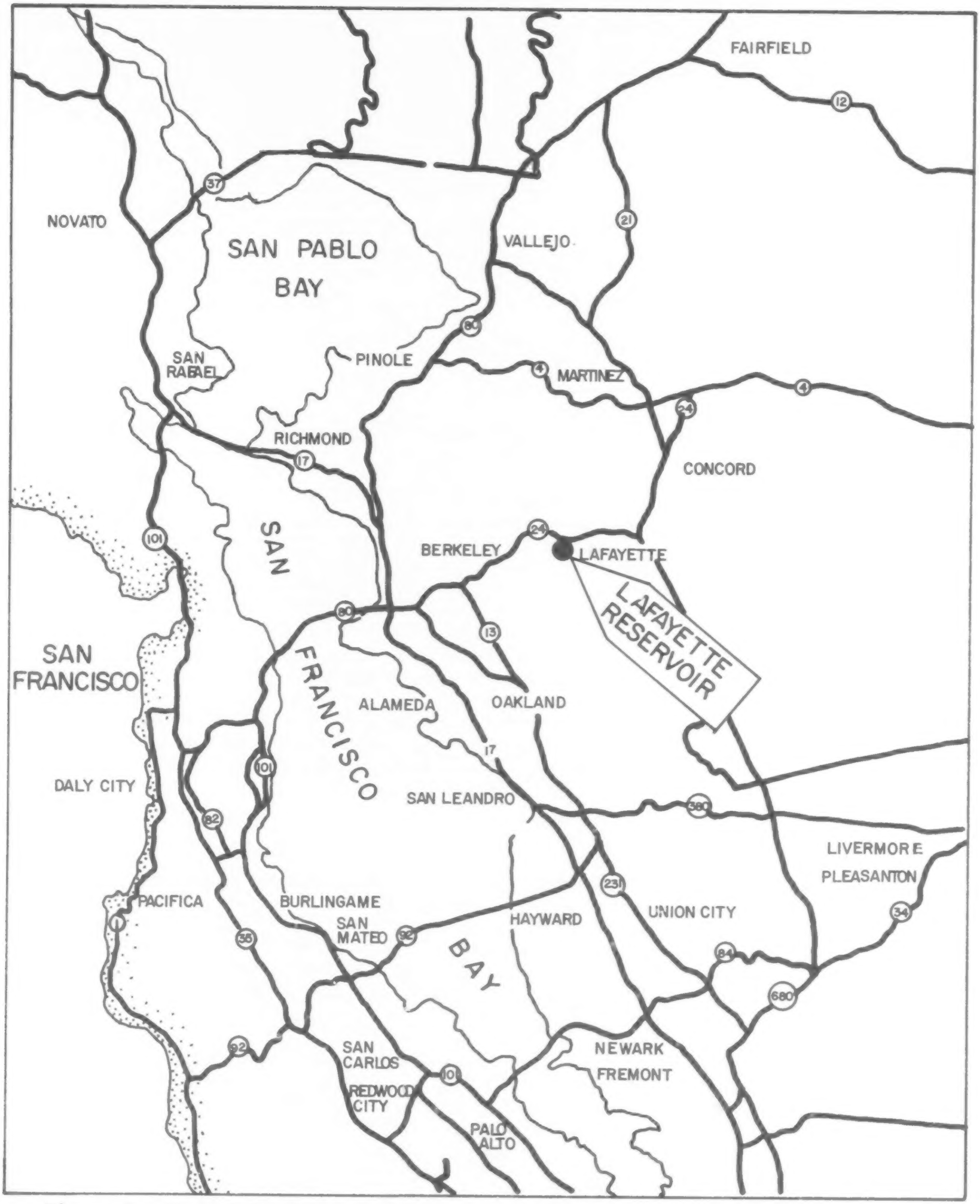
FAIRFIELD
12
37
NOVATO
SAN PABLO BAY
VALLEJO
21
80
SAN RAFAEL
PINOLE
4
MARTINEZ
4
24
RICHMOND
17
CONCORD
SAN
101
24
BERKELEY
LAFAYETTE
LAFAYETTE RESERVOIR
80
13
SAN FRANCISCO
FRANCISCO
ALAMEDA
OAKLAND
17
101
DALY CITY
SAN LEANDRO
580
82
LIVERMORE
PLEASANTON
231
PACIFICA
BURLINGAME
SAN MATEO
92
HAYWARD
UNION CITY
1
35
BAY
34
84
680
92
SAN CARLOS
NEWARK
FREMONT
101
REDWOOD CITY
PALO ALTO

Figure 1. Map of San Francisco Bay area showing location of Lafayette Reservoir.

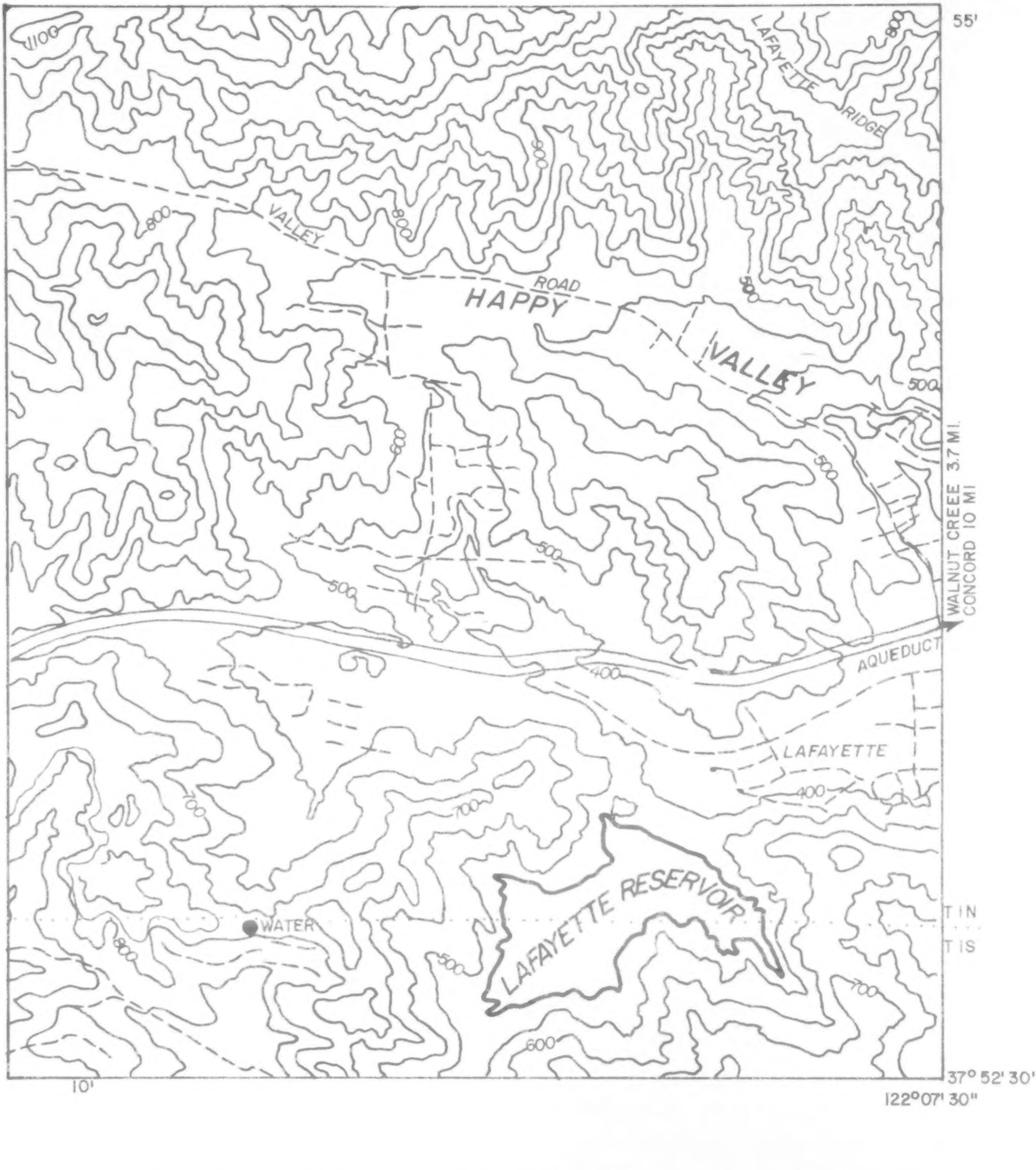
1100
LAFAYETTE RIDGE
800
900
800
VALLEY
ROAD
800
HAPPY
VALLEY
500
500
600
500
500
WALNUT CREEE 3.7 MI.
CONCORD 10 MI
AQUEDUCT
400
LAFAYETTE
400
700
700
LAFAYETTE RESERVOIR
WATER
800
500
700
600
55'
T 1 N
T 1 S
37° 52' 30"
10'
122° 07' 30"

Figure 2. Topographic map of Lafayette Reservoir and neighboring area.

Morphological Characteristics

At maximum capacity, Lafayette Reservoir has a volume of 4,246 acre-feet (5.2×10^6 m^3), a surface area of 128 acres (51 ha) and maximum depth of 80 feet (24 m). The bathymetry of the reservoir is shown schematically in Figure 3. The reservoir volume is generally quite stable. Water levels typically vary less than 5 feet (1.5 m) per year and average about 445 feet (135.7 m) of elevation [approximately 4 feet (1.2 m) below the spillway elevation]. The area-capacity curves presented in Figure 4 show that this elevation corresponds to an average volume of 3,700 acre-feet (4.5×10^6 m^3), an average surface area of 125 acres (50 ha).

Since runoff into the reservoir is very limited, the water level is maintained by importing water from the Mokelumne River which is located in the Central Valley of California. Due to taste and odor problems, Lafayette Reservoir is considered an emergency standby water supply, and little water is withdrawn from it.

Geological and morphological characteristics of the reservoir are summarized in Table 1.

TABLE 1. SUMMARY OF LAFAYETTE RESERVOIR GEOLOGICAL AND MORPHOLOGICAL CHARACTERISTICS

Parameter	Lafayette Reservoir
Location	
Elevation	450 feet
Longitude	122¼ 8' 26" W
Latitude	37¼ 53' 14" N
Drainage Area	830 acres
Evaporation	55 inches/year
Precipitation	26 inches/year
Surface Area	125 acres
Lake Volume	3,700 acre-feet
Depth	
Mean	30 feet
Maximum	80 feet
Epilimnion	30 feet
Length of Shoreline	3 miles
Duration of Stratification	April - November

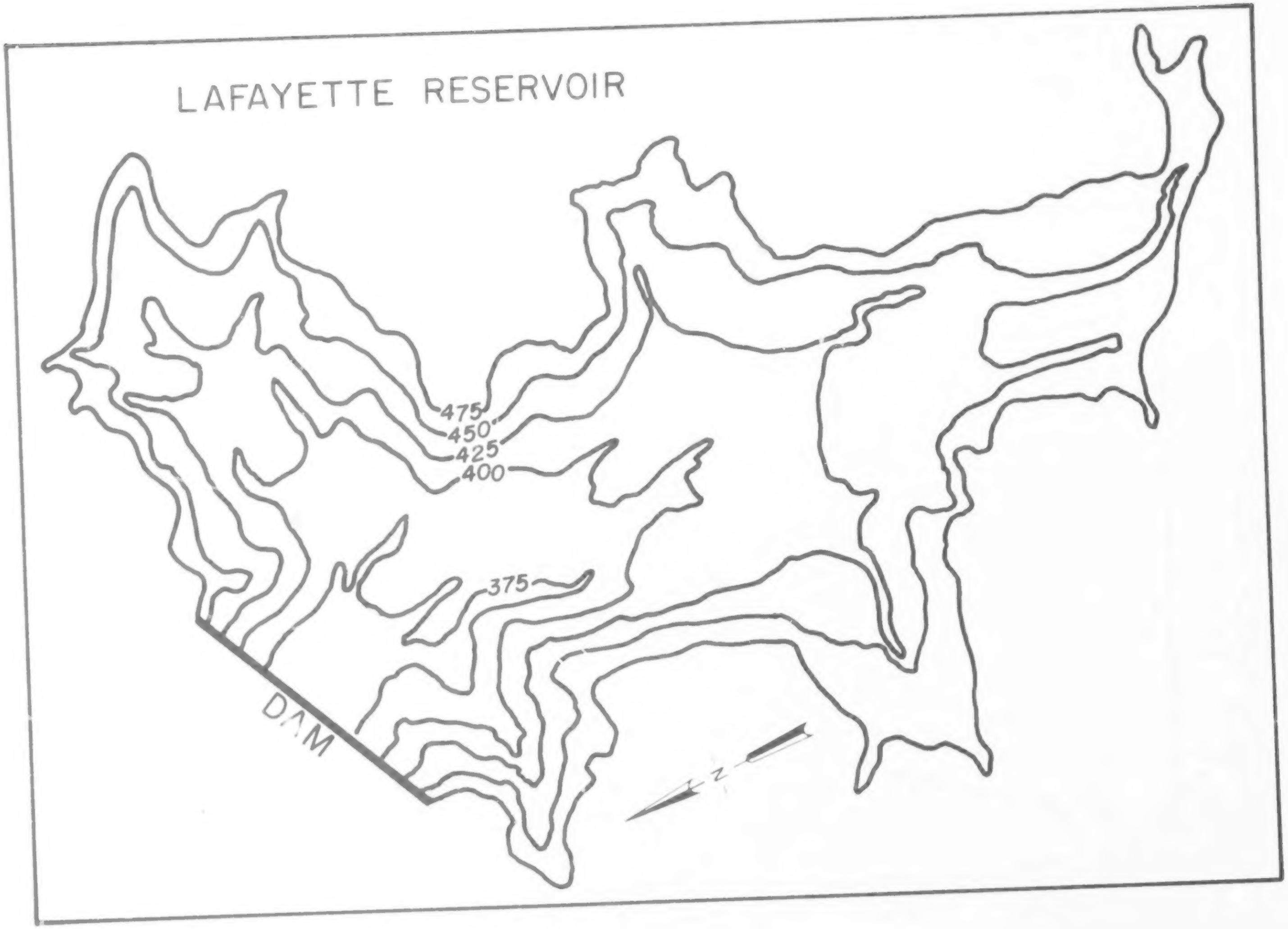
LAFAYETTE RESERVOIR
475
450
425
400
375
DAM
N

Figure 3. Bathymetric map of Lafayette Reservoir basin. Elevations are shown in feet.

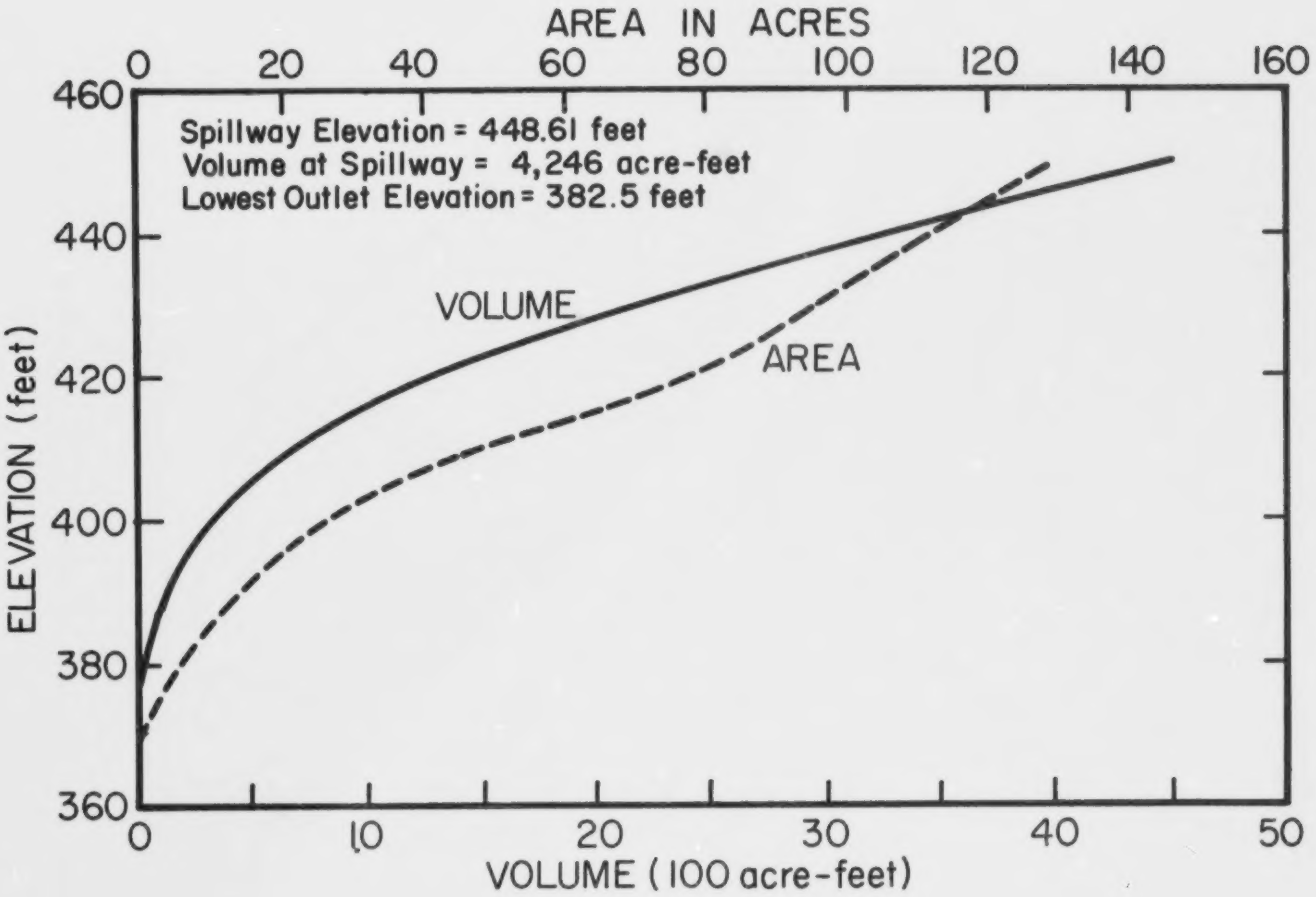

Figure 4. Area - Capacity curves for Lafayette Reservoir (from EBMUD, 1976).

Limnological Characteristics

Lafayette Reservoir is a subtropical, eutrophic lake with sufficient dissolved nutrient to support abundant algal growth. Temperature and oxygen data compiled by EDMUD (1976) show that temperatures range from 8°C to 24°C in surface waters and from 8°C to 14°C near thelake bottom in the deepest part of the lake. Thermal stratification generally begins in March, followed by rapid depletion of hypolimnetic dissolved oxygen. Anoxic conditions typically prevail in the hypolimnion from July through October when the lake destratifies.

In August of 1977, the EBMUD initiated a monthly sampling program to characterize water quality during the pretreatment phase of the restoration project. Samples are taken at depths of 2.5, 5, 10, 15, and 20 meters, and analyzed for temperature, dissolved oxygen (DO), phosphorus, nitrogen, chlorophyll-a, alkalinity, and pH.

Temperature-DO profiles presented in Figure 5 show that stratification was well defined in June of 1977 and continued through November. The thermocline was at a depth of about 30 feet with a maximum ΔT of about 12°C in August. Oxygen concentrations were less than 0.5 mg/l in the deepest part of the lake from June until December. Heavy rains precluded sampling in December, and by January of 1978, the lake was well mixed.

Phosphorus and nitrogen profiles presented in Figure 5 indicate that the major source of nutrients is the organic sediment within the reservoir. Phosphorus concentrations are highest in the hypolimnion during periods of stratification; however, reactive phosphate appears to be well above growth-limiting levels throughout the water column. Nitrogen concentrations are also well above critical levels for algal growth. Ammonia nitrogen predominates in the anoxic hypolimnion, while the euphotic zone is characterized by the more oxidized nitrate and nitrite forms.

While algal growth does not appear to be limited by availability of nutrients, other factors including light and pH extremes do impose some restrictions on algal growth. Profiles of chlorophyll-a, alkalinity and pH presented in Figure 6 show that periods of high algal activity (high chlorophyll and pH) tend to be followed by periods of lower productivity.

The reservoir supports a large growth of blue-green algal species. The EBMUD records indicate that their numbers range from over six million cells per 100 ml in August of 1977 to approximately 2,400 per ml in January of 1978. Green algae are also common, though less abundant.

In order of abundance, the warmwater game fishes include bluegill (Lepomis macrochirus), black crappie (Pomoxis nigromaculatus), white catfish (Ictalurus catus), smallmouth black bass (Micropterus dolomieu), largemouth black bass (M. salmoides), green sunfish (L. cyanellus), and the channel catfish (I. punctatus). Rainbow trout (Salmo gairdneri) are stocked in the lake during the cooler months, but they do not survive through the summer. The fish are stocked at catchable sizes on a "put-and-take" basis. Nongame, warmwater

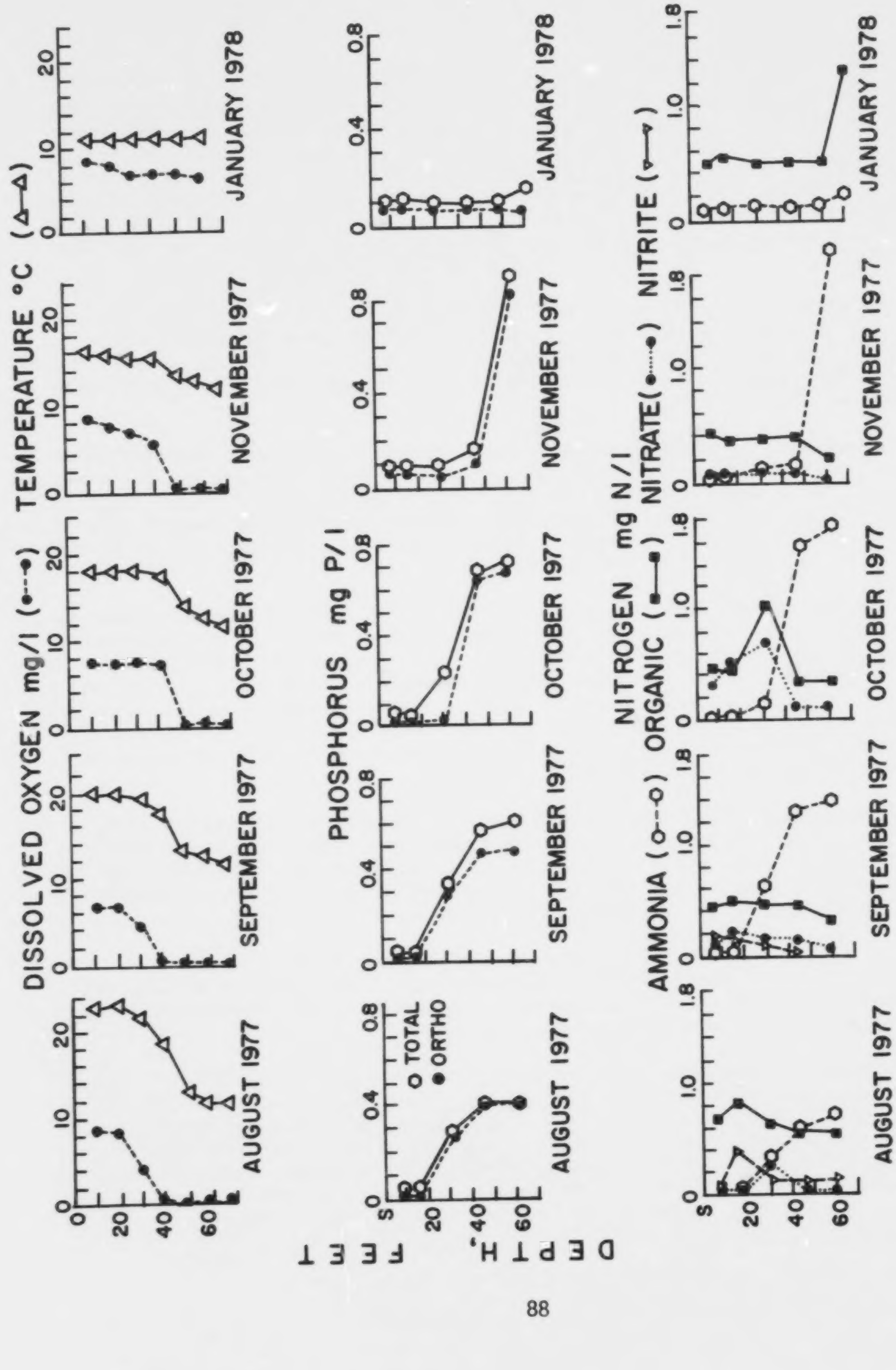

Figure 5. Dissolved oxygen, temperature and nutrient profiles for Lafayette Reservoir.

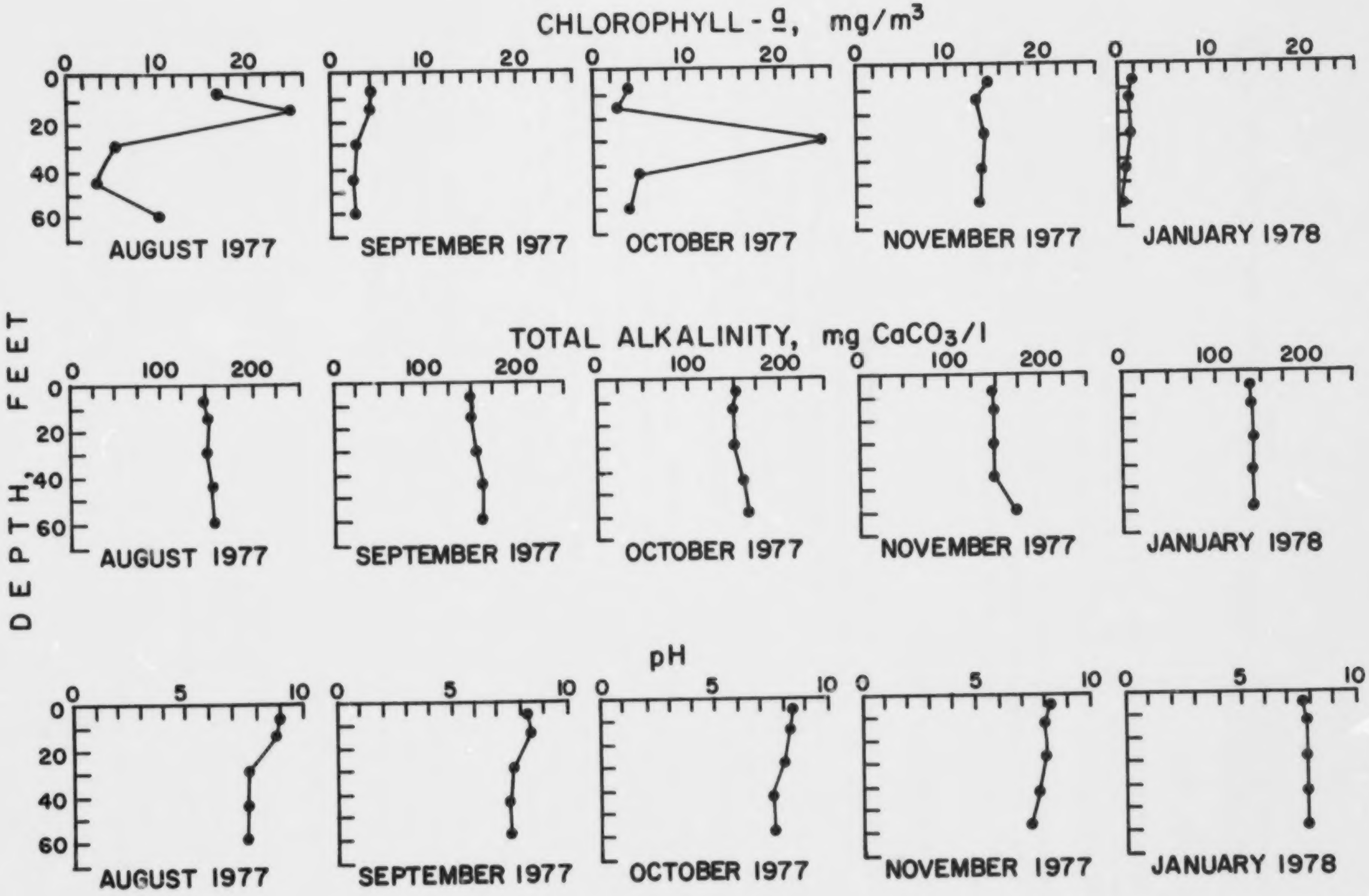

Figure 6. Chlorophyll-a, total alkalinity and pH for Lafayette Reservoir

species include goldfish (Carassius auratus) and Sacramento blackfish (Orthodon microlepidotus)(EBMUD, 1976).

Recreational Uses

Lafayette Reservoir was first opened for recreational use in 1966. Total visitation per year has grown from 127,000 during fiscal year 1967-1968 to 341,000 during 1974-1975, and is expected to continue to increase.

Recreational facilities at Lafayette Reservoir include fishing, boating, picnicking, bicycling, and hiking. There are over nine miles of hiking trails around the lake and surrounding park and paved bicycle paths along the three-mile lake perimeter.

PROPOSED RESTORATION PROGRAM

The EBMUD proposes to install a hypolimnetic aerator similar to the device described by Lorenzen and Fast (1976) which is shown schematically in Figure 7. The intent is to provide a suitable habitat for cold water game fish by aerating the hypolimnion while maintaining thermal stratification of the lake. The aerator will be placed at or near the deepest portion of the lake and is expected to be operational by July of 1978. The system will be operated seasonally during periods of thermal stratification.

In addition to hypolimnetic aeration, the reservoir will be treated with alum (aluminum sulfate) twice during the summer of 1978. Alum has been used as an in situ nutrient inactivation procedure by several investigators (Dunst, 1974; Cooke and Kennedy, 1977; Funk, et al., 1977; Barrion, 1976) and has been shown to be an effective phosphorus removal process. Layers of alum from 1-2 cm thick have been observed to form at the sediment-water interface. This layer can be an effective phosphorus trap to prevent release of phosphorus from the sediment.

Alum will be applied to the surface water (about 70 tons) during the summer and to the hypolimnion (about 130 tons) in the fall. Since the primary source of nutrients in Lafayette Reservoir is internal, it is believed that these treatments should significantly reduce nutrient regeneration, and therby improve the quality of water in the reservoir.

PROPOSED MONITORING PROGRAM

Tetra Tech will undertake a supplemental monitoring program to measure chemical and biological characteristics before, during, and after restoration. Physical properties of the system will also be identified in order to estimate water, nutrient, and DO budgets.

Water samples will be collected weekly in the summer (June, July, August and September) and monthly otherwise. Water quality parameters such as pH, temperature, and transparency will be measured in the field. Whole water samples will be taken at several depths and analyzed for nutrients, chlorophyll-a, pH, and alkalinity.

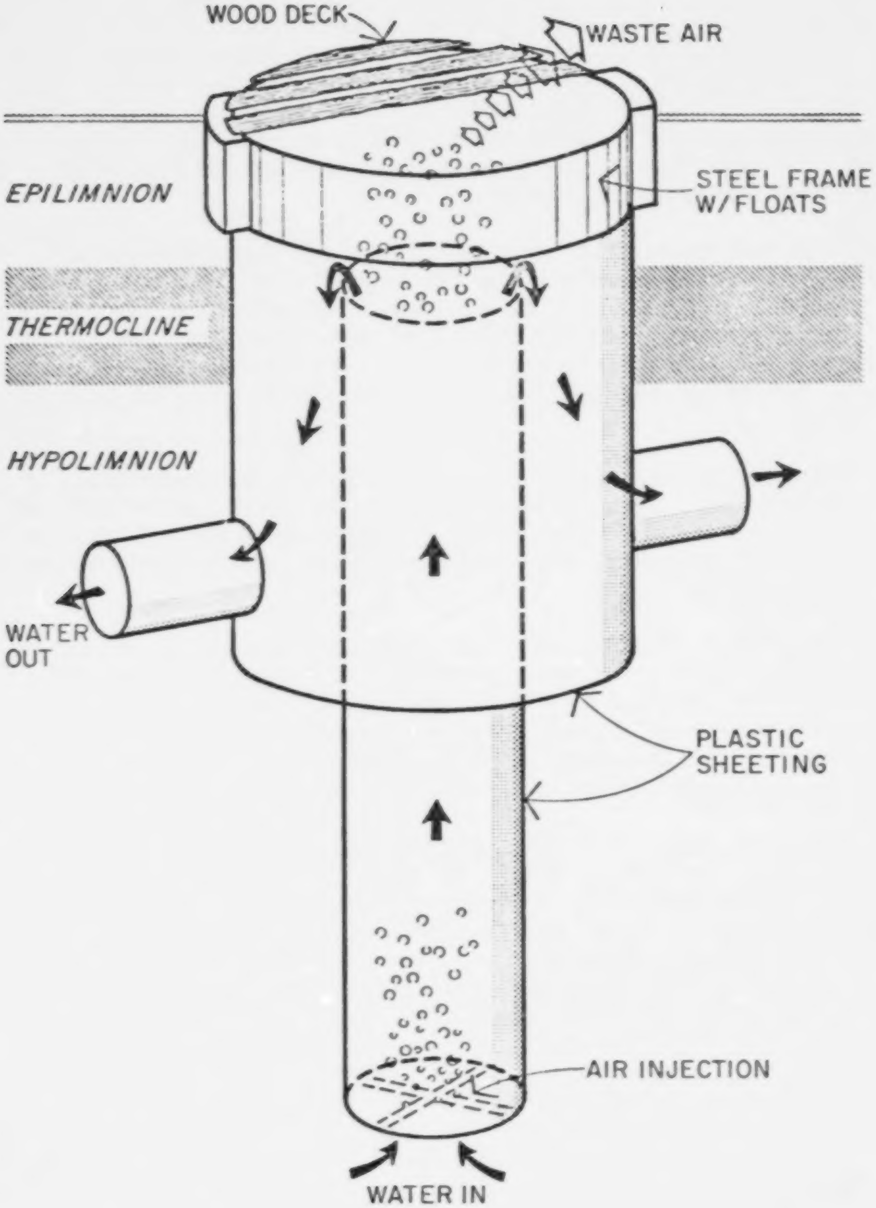
WOOD DECK
WASTE AIR
EPILIMNION
STEEL FRAME W/FLOATS
THERMOCLINE
HYPOLIMNION
WATER OUT
PLASTIC SHEETING
AIR INJECTION
WATER IN

Figure 7. Hypolimnion aerator by Fast (1971).

Sediment chemistry studies will be conducted to determine the sediment characteristics before and after alum treatment. Core samples will be taken from three locations, one before treatment and four times at five-month intervals following treatment. Samples will be sectioned into at least three layers. Parameters which will be measured include percent organic, total phosphorus, iron, copper and aluminum. In addition, several sediment grab samples (before and after treatment) will be incubated in the laboratory for long-term determination of total exchangeable phosphorus. These tests will be conducted in a semi-continuous fashion by decanting and replacing a portion of the supernatant approximately weekly.

Nitrogen and phosphorus release rates from the sediments will be studied in situ with a lucite chamber. At the beginning of the experiment, the chamber is placed on the station by a diver. Periodically (hourly, bihourly or daily), the diver will extract 50 ml of water for the analyses of pH, dissolved oxygen and nutrient concentrations. The time series of results will be plotted to determine the benthic oxygen demand as well as nutrient exchange rates.

In addition to measuring internal recycling of nutrients, a survey of external sources will also be made. Samples of runoff water will be collected at three locations, five times each, during five different rainfall periods. Concentrations of nutrients in controlled water inflows will be obtained from EBMUD.

Benthic animal samples will be collected quarterly at three stations. Species will be identified and quantified to the extent possible. The littoral zone of the reservoir will be examined semiannually for the presence of aquatic macrophytes. If found, these plants will be identified and a description of location and abundance will be provided in order to monitor possible increases in macrophyte growth as a result of improved water transparency.

Phytoplankton and zooplankton populations will be monitored weekly during the summer, an on a monthly basis for the remainder of the sampling period. Samples will be collected by a discrete sampler (phytoplankton) and pump-set (zooplankton) from at least two stations at two depths. Species enumeration for phytoplankton and zooplankton will be made for predominant organisms. The reservoir plankton will be analyzed for seasonal population trends, species composition and depth distribution.

Aeration of the hypolimnion should provide a suitable habitat for year-round trout survival. Because a possible improved cold water fishery could provide a significant recreational benefit, a fishery survey would be conducted. A trout tagging study based upon the return of tags from fishermen-caught tagged fish will be undertaken to determine the survival time of the reservoir trout population.

Creel census data will also be used to measure angler use, fishery preference, and catch per hour. The creel census will involve a voluntary participation and will utilize questionnaires given to reservoir visitors. These questionnaire surveys will be conducted weekly during the summer. The creel census will be for the following items: number of people fishing, start time, stop time, fishing from boat or shore, what species fishing for, and the number of each species caught and kept. Information gained would include rates of fishing success for each species over a long period of time for boat and shore fishermen, periods of greatest activity and success rate, and fishing preferences.

The distribution of fish in the Reservoir will be examined with vertical gill nets. Two adjacent gill nets of different mesh sizes will be fished at three sampling stations. The nets will be of sufficient length to extend from the surface to the bottom. Upon retrieval, the size and position in the net will be recorded for each captured fish. Stomach contents of captured fish will also be analyzed. One gill net station will be located in close proximity to the hypolimnetic aerator. The remaining two stations will be positioned at increasing distances from the aeration point. The gill net samples will provide direct information on fish utilization of hypolimnetic habitat both before and after aeration.

The shallow-water fish populations of Lafayette Reservoir will also be characterized before and after hypolimnetic aeration. The primary sampling device used will be a 50-foot beach seine.

Live cages suspended in the hypolimnion following the initiation of aeration will be used to analyze the suitability of aeration bottom waters as trout habitat. Live cages will be positioned at a station adjacent to the aerator, and also at a minimum of one station located at selected increasing distances from the aeration point. Trout survival in the cages will be monitored at selected intervals by diver observation.

Additional information which will be compiled for the study includes rainfall data, groundwater data, evaporation rates, and data for physical variables including dispersion and advection. Rainfall will be accurately recorded with a rain gauge at the site. Groundwater flow will be determined at three stations in conjunction with the sediment chemistry program. Evaporation rates will be computed from field measurement of pan evaporation and compared with rates computed by the ecological model which will be applied to the reservoir.

Lake bathymetry and direct inflow data will be obtained from EBMUD together with weekly readings of lake level. Surface runoff will be computed by difference utilizing rainfall data.

SYSTEM EVALUATION

The first step in system evaluation is to determine the mechanisms for water quality improvement. Alum treatment and hypolimnetic aeration can act synergistically. Alum treatment may reduce the phosphorus release rate from the organic sediment so that the wintertime phosphorus concentration is lowered. Lowered wintertime phosphorus concentration may support lower summertime standing crops of algae which may in turn consume less oxygen from the hypolimnion water. This may reduce the need for hypolimnetic aeration. Also, the decreased algal standing crop may increase the light penetration and modify the thermal structure of the lake water. Hypolimnetic aeration should keep the hypolimnion water aerobic throughout the year. Depending on the chemical forms, phosphorus release rates may be decreased. This may in turn reduce the wintertime phosphorus concentrations and the summertime phytoplankton density.

Computer modeling techniques will be used to help clarify the mechanisms involved in the lake response to treatment and to provide a predictive, analytical tool. Two levels of modeling will be used: (1) detailed simulation, and (2) longer term nutrient budget analysis.

The water quality ecological model represents the lake by a series of layers. Heat budget and mass balance computations are performed to calculate the water quality profiles for temperature, pH, DO, nutrients (P, N, C), phytoplankton (4 groups) and zooplankton (2 groups). Simulations are performed throughout the annual cycle, usually with a daily time step. The basic principles and formulations of the model have been well documented (2). The model has been modified to evaluate the effects of hypolimnetic aeration. Further modification will be made to include the effects of alum treatments that may remove phosphorus and particulate matter from the water column and also reduce the rate of phosphorus release from the sediment. It is not certain if the change in sediment characteristics will reduce the decay rate of the organics and therefore the oxygen depletion rate in the hypolimnion. The model computes oxygen dynamics based on physical variables (temperature, mixing, advection, gas exchange) and biochemical processes (algal respiration, detritus decay, sediment oxygen demand). These data will be used together with information pertaining to aerator performance to analyze the oxygen budget over time.

In addition to detailed simulation modeling, a nutrient budget model such as applied to Lake Washington by Lorenzen *et al*. (1976), will be used in the analysis of the nutrient budgets. This model is based on a mass balance which considers loading from all sources, loss to the sediments, release from the sediments and discharge (if any). The model can be operated in a dynamic or steady-state mode. Concentrations of nutrient in both the water and sediment are simulated. For Lafayette Reservoir, it is expected that sediment exchange will be a critical process. As pointed out by Lorenzen *et al*. (1976), the sediment nutrient release rate constant should not affect long-term, steady-state water concentrations. However, it may have a marked influence on short-term fluctuations which could be important during the growing season.

TIME SCHEDULE

The work schedule will be closely coordinated with EBMUD. Preoperational studies are under way and will continue until the aerator is operational (July, 1978). The program will continue for a period of two years and will be completed in 1980.

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LIMNOLOGICAL CHARACTERISTICS OF LONG LAKE
KITSAP COUNTY, WASHINGTON

by

M. A. Perkins, E. B. Welch and J. O. Gabrielson*

INTRODUCTION

Long Lake, Kitsap County, Washington, has been selected as one of several lakes in the United States to be rehabilitated from a eutrophic state. The lake has shown considerable algal blooms during the spring and summer months, and has extensive macrophyte beds, particularly *Elodea densa*, which reach nuisance proportions.

The proposed rehabilitation measures consist of stormwater treatment, lake drawdown during the summer, alum addition, and limited dredging. The implementation of these treatment measures will be under the direction of Entranco Engineers, Bellevue, Washington. The objectives of this study are to evaluate selected limnological characteristics which reflect the condition of the lake prior to, during, and after the application of the designated treatments. Emphasis has been placed upon inorganic nutrient interactions, particularly phosphorus. The results to date represent the pretreatment phase of the rehabilitation effort as the restorative techniques have yet to be implemented. Application of treatment measures is scheduled to begin with lake drawdown during the summer, 1978.

The purpose of this report is to summarize the data collected over the period July, 1976 to December, 1977.

DESCRIPTION OF STUDY AREA

Long Lake, Kitsap County, is a long (2.8 km) narrow (0.25 km) relatively shallow lake located in the Puget Sound Basin, near the City of Port Orchard, Washington (T23N-R2E-SEC 17). The drainage area for the lake is approximately 24.3 km^2 (9.36 sq. miles), most of that area being forest or undeveloped land (69%). Approximately 5% of the drainage basin is classified as residential suburban, with 121 near shore homes (USGS, 1973). Public access to the lake is provided by a boat ramp located in the vicinity of Salmonberry Creek and a boat rental concession located along the eastern shore. Recreational uses of the lake include sport fishing, boating, and swimming.

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The lake has a single outlet, Curley Creek, at the north end. The major inflow is Salmonberry Creek, with other less significant inflows draining, primarily, into the southern end of the lake. The lake surface area of 137 hectares (340 acres) represents approximately 6% of the drainage basin.

As indicated, the lake is relatively shallow with a maximum depth of 3.7 meters (12 feet). Approximately 72% of the lake surface covers waters less than 3 meters (10 feet) in depth and 28% is less than 1.5 meters (5 feet). The basic morphometric features of Long Lake are presented in Figure 1.

METHODS AND MATERIALS

BASIN HYDROLOGY

The hydrologic features of the Long Lake basin have been monitored primarily by Entranco Engineers. Calibrated stage level recorders have been installed on Salmonberry Creek, the major inflow, and Curley Creek, the outlet, for continuous monitoring of discharge. Additional inflows to the lake have been estimated to be 17% of Salmonberry Creek and direct surface runoff as 24% of Salmonberry. These estimates were based upon precipitation and drainage characteristics of the basin (Entranco data). Precipitation inputs were based upon data measured and recorded at the Kitsap County Airport.

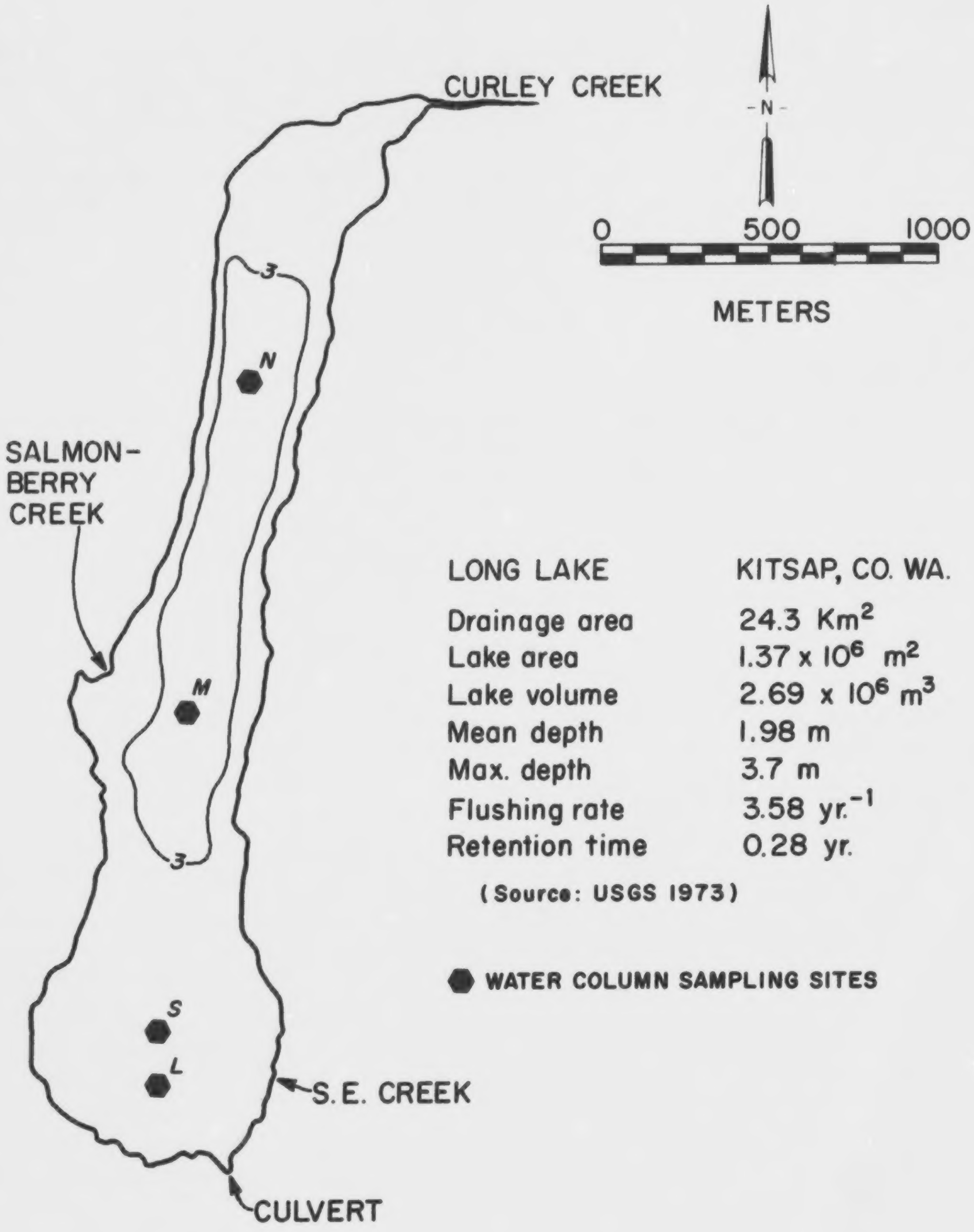
Ground water inputs were estimated using the Minnesota half-barrel technique (Lee, 1977). Thirteen half-barrel seepage meters were placed at various locations within the lake. Nine of these seepage meters were placed along the northeast shore in an attempt to evaluate possible influences of septic tank drainage from the concentration of homes along that shore.

WATER QUALITY CHARACTERISTICS

Samples for chemical analysis were collected from the inlet streams (Salmonberry, S.E. Creek, S.E. Culvert), 4 lake stations, the ground water seepage meters and the Curley Creek outlet. Sampling frequency was weekly during the late spring to early fall period and biweekly during the winter. The sampling effort on the inflow and outflow creeks was divided between UW and Entranco personnel such that sampling on alternate weeks gave nearly weekly observations on these creeks for the whole period of investigation. Sampling at the lake stations occurred at three depths; surface, mid-depth, and bottom for the north, midlake, and south stations while surface samples only were taken at the southern most, lillius, station. Rainfall samples at the lake were collected and analyzed by Entranco personnel.

The chemical analyses on the collected samples are listed by parameter and location in Table 1. Standard Methods (APHA, 1971) was followed for each parameter listed.

Biological measurements at the lake stations included chlorophyll a (Flurometric determination, Strickland and Parsons, 1972) and primary production (^{14}C uptake, Strickland and Parson, 1972). Measurements of inorganic carbon available for production were made by direct determination using infarared gas analysis (Perkins, unpublished).


CURLEY CREEK
N
0
500
1000
METERS
3
N
SALMON-
BERRY
CREEK
M
3
S
L
S. E. CREEK
CULVERT
WATER COLUMN SAMPLING SITES

LONG LAKE	KITSAP, CO. WA.
Drainage area	24.3 Km^2
Lake area	1.37×10^6 m^2
Lake volume	2.69×10^6 m^3
Mean depth	1.98 m
Max. depth	3.7 m
Flushing rate	3.58 $yr.^{-1}$
Retention time	0.28 yr.

(Source: USGS 1973)

Figure 1. Water column sampling locations and morphometric characteristics of Long Lake.

TABLE 1. CHEMICAL ANALYSIS OF LONG LAKE SAMPLES BY PARAMETER AND LOCATION

Parameter	Location				
	Creeks	Lake	Sed.	GW	Precip.
Tot-P	X	X	X	X	X
PO_4-P (SRP)	X	X			
Tot-N	X	X		X	
NO_2+NO_3-N	X	X			
NH_3-N	X	X			
pH	X	X			
Alkalinity	X	X			
D.O.		X			

Measurements of Secchi depth and temperature were also made at the lake stations.

SEDIMENTATION

Sedimentation rates in Long Lake were measured using an array of collecting tubes fastened to the end of 10 cm funnels held in a plexiglass frame. The frames, each holding four collecting funnels, were suspended in the water column at the midlake station. The depths of suspension were 0.3 and 1.0 meters above the bottom. In order to correct for factors of sediment resuspension, a double tiered design was also placed in the lake. Samples from the sediment collectors were taken at 4 week intervals and the dry weight and total phosphorus content was determined (Gabrielson, 1978).

During the course of this investigation, questions were raised relating to the history of sedimentation in the lake and the influence of resuspension upon the measured sedimentation rates (as outlined above). In order to address these questions, two 30 cm sediment cores were taken for geochronological dating, using stable lead, stable aluminum, and phosphorus concentrations in the sediment profile. Stable lead and aluminum concentrations were determined by atomic absorption after digestion with HF-HNO_3-$HClO_4$. Total phosphorus was determined as molybdate reactive phosphate after the digestion.

Dating of the sediment profile was accomplished by relating the measured concentrations to the cultural and fluvial history of the Puget Sound basin. An additional cross check of the dates established was made using cesium-137 activity in the profile. The procedures used in the geochronological dating were those of Schell and Barnes (1974).

MACROPHYTE SURVEY

Macrophyte biomass estimates were made within three major areas of the lake. Designation of the sampling areas was based upon qualitative estimates of plant distribution, characteristics of the sediment substrate, and water depth. These areas were: (1) a shallow water south area having a homogeneous muck substrate and uniform plant density; (2) a deep water midlake area having a fairly homogeneous muck substrate and scattered distribution of plants; and (3) a shallow water north area having a heterogeneous substrate type and plant distribution. The three sampling areas comprised approximately 30, 59 and 11 percent of the lakes surface, respectively (Figure 2).

Samples were collected in a steel cylinder, one end of which was covered with fish netting to prevent the loss of plant materials. The cylinder enclosed an area of 0.255 m^2 when placed into the lake bottom. Plant materials within the enclosed area were removed by divers and returned to the laboratory for species identification and determination of dry weight biomass (Gabrielson, 1978). Samples of Elodea densa, the dominant macrophyte in Long Lake, were further analyzed for total phosphorus content after ashing and nitric acid digestion (Chapman and Pratt, 1961). E. densa collected from Long Lake was also grown in the laboratory using Long Lake sediments as the rooting media. These laboratory grown plants were used in radiotracer studies to follow patterns of uptake and translocation of phosphorus. The details of the radiotracer experiments will appear elsewhere (Gabrielson and Perkins, in preparation).

Samples for biomass determination were taken in September, 1976 at 44 lake stations and in October, 1977 at 23 lake stations.

RESULTS

WATER BUDGET

The stage level recorders were installed on Salmonberry and Curley Creek in October, 1976 and a continuous record of discharge has been kept since that time. The yearly discharge data for these creeks are presented in Figure 3. The data points are summarized as 5-day totals for ease of presentation and cover the period October, 1976 to October, 1977. The average flow rates over the period were 19 $m^3 \cdot min^{-1}$ (11 cfs) for Curley Creek and 11 $m^3 \cdot min^{-1}$ (6 cfs) for Salmonberry Creek. Peak flows in both creeks were observed in March, 1977 with maximum rates of 156 $m^3 \cdot in^{-1}$ (92 cfs) for Curley Creek and 143 $m^3 \cdot min^{-1}$ (84 cfs) for Salmonberry. Minimum flows of 1.7 $m^3 \cdot min^{-1}$ (1 cfs) were observed in both creeks during August, 1977.

The total water input to the lake through Salmonberry Creek was 5.90 x 10^6 m^3 (4782 acre-feet) and that leaving the lake through the Curley Creek outlet was 9.64 x 10^6 m^3 (7813 acre-feet).

Precipitation data, total centimeters per month, are presented in Figure 3.

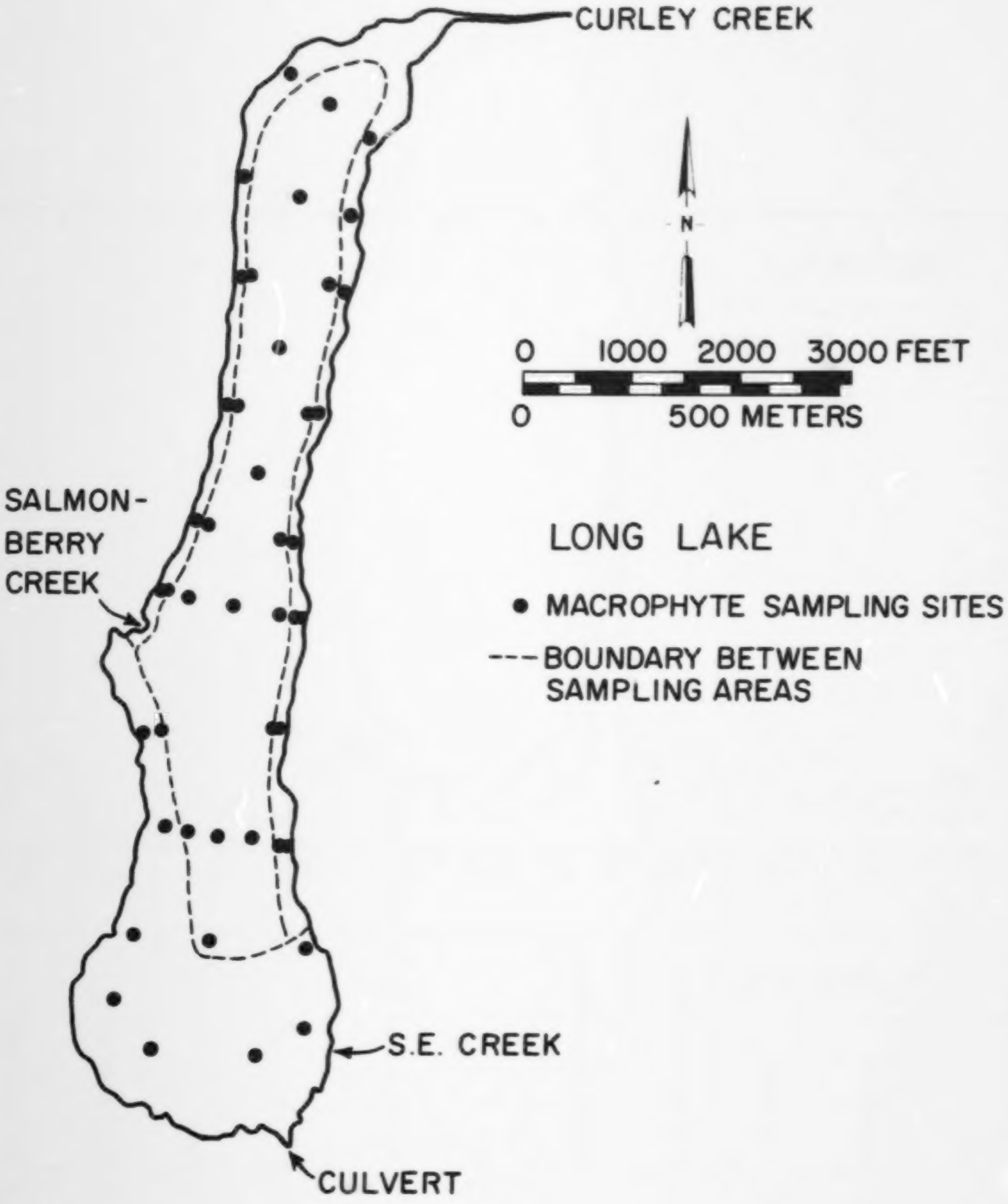

CURLEY CREEK
N
0 1000 2000 3000 FEET
0 500 METERS
SALMON-BERRY CREEK
LONG LAKE
● MACROPHYTE SAMPLING SITES
--- BOUNDARY BETWEEN SAMPLING AREAS
S.E. CREEK
CULVERT

Figure 2. Map of Long Lake showing macrophyte areas and the locations of actual sampling sites.

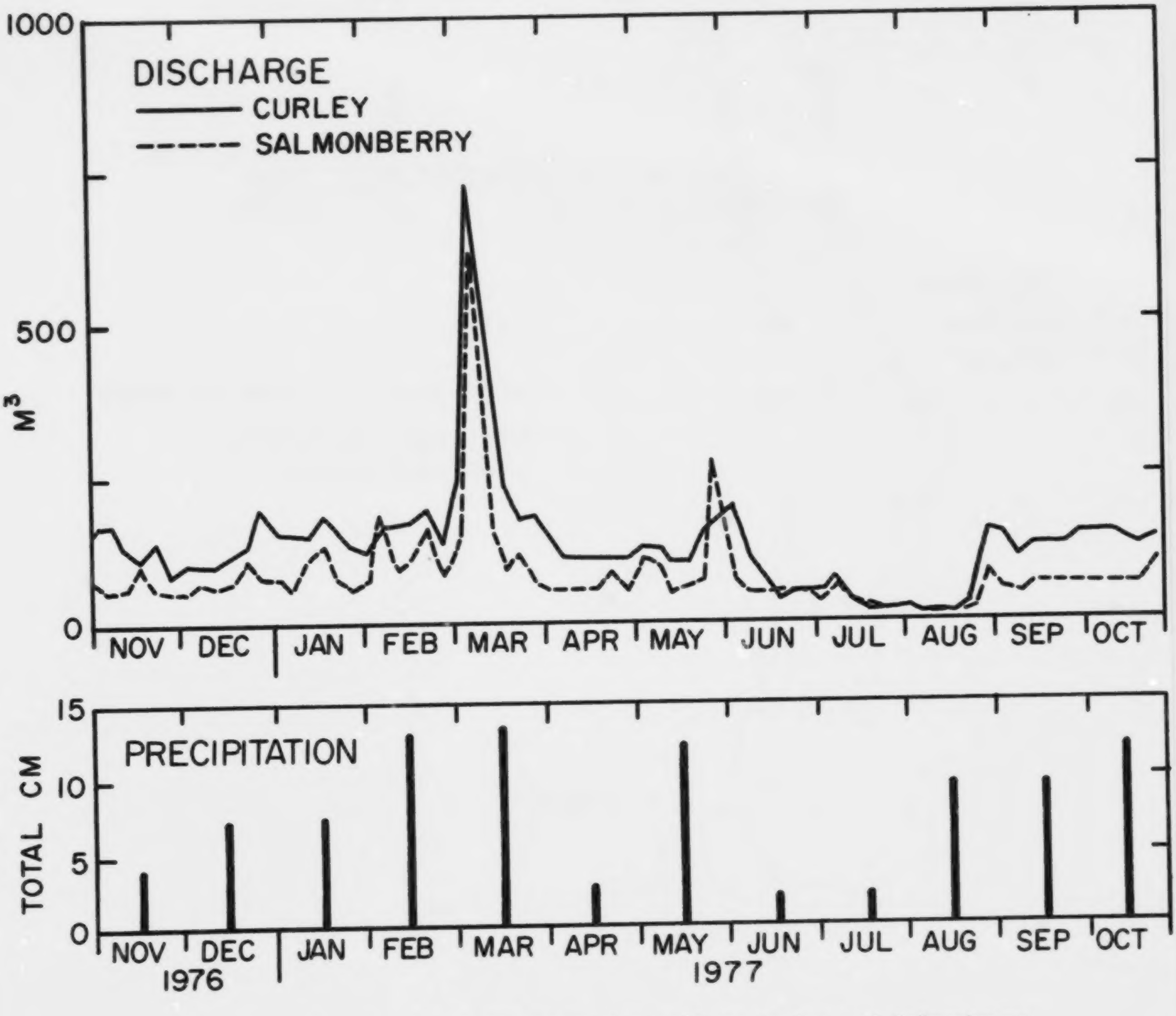


Figure 3. Five day total discharge (meter³) in Curley and Salmonberry Creeks for the period October, 1976 to October, 1977. Total precipitation (cm) measured at Kitsap County airport.

The total precipitation for the period October, 1976 to October, 1977 was 97.21 centimeters (38.3 inches). Approximately 42% of the total precipitation fell during the period December, 1976 to March, 1977. The direct input to the lake over the total period was 1.33 x 10^6 m^3 (1079 acre-feet).

Estimates of ground water input were obtained from the Minnesota half-barrel seepage meters. Flow rates from the seepage meters varied over a range of 0.006 to 0.60 m^3 min^{-1} (.004 to 0.35 cfs). Measurements of ground water flow were averaged on a quarterly basis (3 months) and the quarterly inputs were obtained by multiplying the quarterly daily averages by the number of days in the quarter. The total input for the period October, 1976 to September, 1977 was taken as the sum of the quarterly inputs and amounted to 1.6 x 10^5 m^3 (130 acre-feet) or 2% of the total inflow.

Curley Creek represented the dominant water loss from Long Lake. Evapotranspirative losses were estimated as ten percent of the total output.

A summary of the Long Lake water budget for the period October, 1976 to October, 1977 is presented in Table 2.

TABLE 2. LONG LAKE WATER BUDGET, OCT. 27, 1976 - OCT. 26. 1977

Inputs:			
	Salmonberry Creek	5.9 x 10^6	60
	*other creeks	1.0 x 10^6	10
	*run off	1.4 x 10^6	14
	precipitation	1.3 x 10^6	13
	ground water	1.6 x 10^5	2
		9.76 x 10^6	100
Outputs:			
	Curley Creek	9.6 x 10^6	90%
	evapotranspiration	1.1 x 10^6	10%
		10.7 x 10^6	100%

* Other creeks estimated at 17% of Salmonberry Creek surface; runoff estimated at 24% of Salmonberry Creek; estimates from ENTRANCO data.

STREAM NUTRIENT CONTENT

A summary of the inorganic nutrient concentrations in the Salmonberry Creek inlet and Curley Creek outlet are presented in Table 3. The values reported are quarterly mean concentrations ± two standard errors as an approximation of the 95% confidence interval about the mean.

TABLE 3. QUARTERLY MEAN CONCENTRATIONS (± 2 STANDARD ERRORS) FOR SELECTED WATER QUALITY CHARACTERISTICS FOR THE LONG LAKE WATERSHED

Parameter	Quarter					
		1976	1977			
	Jul - Sep	Oct - Dec	Jan - Mar	Apr - Jun	Jul - Sep	Oct - Dec
Sample days in period	7	6	4	7	6	4
Lake Stations						
TOT-P ($\mu g \cdot l^{-1}$)	39.1 ± 6	33.9 ± 5	43.8 ± 7	43.5 ± 3	67.1 ± 9	44.5 ± 10
PO_4-P "	4.1 ± 0.9	8.1 ± 1.7	4.7 ± 0.5	6.9 ± 2.1	7.9 ± 2.6	10.1 ± 2.1
TOT-N "	519 ± 38	278 ± 41	594 ± 83	717 ± 67	821 ± 63	772 ± 141
NO_2+NO_3-N "	3.4 ± 1.2	37.3 ± 16	130 ± 27	40.3 ± 21	17.3 ± 12	326 ± 153
NH_3-N "	---	12.7 ± 1.4	13.4 ± 3.6	30.3 ± 10	30.4 ± 11	37.8 ± 13
DO ($mg \cdot l^{-1}$)	7.8 - 11.1	6.5 - 12.5	10.3 - 11.6	7.4 - 11.8	4.9 - 12.2	6.7 - 10.9
Temp. (°C)	22.7 - 17.6	16 - 5.5	6.0 - 10	13 - 20.2	25 - 14	13.1 - 2.6
Chla ($\mu g \cdot l^{-1}$) (means of four water column stations)	10.2 ± 3	7.0 ± 4	35.4 ± 18	11.7 ± 4	29.1 ± 9	9.0 ± 3
Salmonberry Ck.						
TOT-P ($\mu g \cdot l^{-1}$)	51.1 ± 8	39.7 ± 3	47.1 ± 21	52.2 ± 17	51.1 ± 7	61.5 ± 43
PO_4-P "	16.6 ± 1.2	13.2 ± 18	12.2 ± 1.2	14.6 ± 2.5	19.7 ± 2.7	11.8 ± 4.1
TOT-N "	492 ± 132	326 ± 60	1020 ± 652	637 ± 85	618 ± 165	1004 ± 335
NO_2+NO_3-N "	185 ± 38	257 ± 119	645 ± 251	204 ± 56	207 ± 21	587 ± 345
NH_3-N "	---	14.3 ± 4	38.6 ± 36.4	16.8 ± 4.5	34.5 ± 18	29.1 ± 14
Other Cks.						
TOT-P ($\mu g \cdot l^{-1}$)	63.6 ± 20	38.2 ± 5	30.5 ± 7	39 ± 4	46 ± 10	49.4 ± 40
PO_4-P "	22.7 ± 1.9	18.1 ± 1.8	16.6 ± 1.7	20.1 ± 2.2	23.7 ± 2.7	13.8 ± 3.1
TOT-N "	297 ± 161	242 ± 94	688 ± 265	559 ± 156	445 ± 160	990 ± 348
NO_2+NO_3-N "	203 ± 132	190 ± 94	455 ± 195	221 ± 80	241 ± 116	676 ± 317
NH_3-N "	---	10.1 ± 1.8	21 ± 12.4	24.9 ± 13.6	24.6 ± 6	22.7 ± 7
Curley Ck.						
TOT-P ($\mu g \cdot l^{-1}$)	45.1 ± 5	33.0 ± 7	38.8 ± 16	[illegible]6.3 ± 5	70.5 ± 8	67.5 ± 46
PO_4-P "	9.7 ± 0.8	9.8 ± 2.2	4.6 ± 1.1	8.1 ± 3.5	10.5 ± 4.0	12.6 ± 1.7
TOT-N "	418 ± 77	189 ± 116	545 ± 272	641 ± 86	800 ± 116	921 ± 203
NO_2+NO_3-N "	19.2 ± 13	27.2 ± 8	160 ± 63	59.4 ± 31	27.3 ± 26	414 ± 312
NH_3-N "	---	12.1 ± 3.4	14.7 ± 3.5	17.6 ± 6.2	37 ± 20	32.6 ± 10

DO = range for bottom samples during period.
Temp = range for surface samples during period.

A more detailed presentation of the data for total phosphorus and total nitrogen is given in Figure 4. Clearly, the variation in concentration makes the discussion of seasonal trends somewhat tenuous.

The average concentration of total phosphorus over the eighteen month period was fairly comparable in both Curley and Salmonberry Creeks, 49 and 50 mg m^{-3}, respectively. The range in concentration was 25.5 to 112.6 mg m^{-3} for Curley Creek and 29.7 to 104.7 mg m^{-3} for Salmonberry Creek. While the average concentrations in the inlet and outlet were comparable over the entire period it was also evident that inlet concentrations were generally greater than outlet concentrations with the exception of the summer and fall period of 1977 (average outlet concentration of 70.5 mg m^{-3} versus an average inlet concentration of 51.1 mg m^{-3}).

Concentrations of total nitrogen showed a marked increase in 1977. The average concentrations over the eighteen month period were 642 mg m^{-3} for Salmonberry Creek (range of 194 to 1922 mg m^{-3}) and 569 mg m^{-3} for Curley Creek (range of 40 to 1133 mg m^{-3}). The peak concentrations in Salmonberry Creek were observed in March, 1977 during the period of maximum discharge (such was not the case with total phosphorus). As with total phosphorus, the inlet concentrations were generally greater than the outlet concentrations, again with the exception of the July to September period. For this period the average outlet concentration was 800 mg m^{-3} versus 618 mg m^{-3} for the inlet. Of interest is the observation that while N concentrations increased abruptly with flow increase in February-March, no such associated increase in phosphorus occurred (Figure 4).

It was also evident that the concentrations of both total-P and total-N over the summer and fall periods of 1977 were greater than those occurring for the comparable period of 1976. This probably reflects the fact that the fall of 1976 was much drier than the fall of 1977. The very dry winter of 1976-77 may have resulted in less dilution of the lake nutrients and allowed even higher buildups from internal sources during July-September 1977. Note the much higher outflow than inflow concentration at that time.

WATER COLUMN CHARACTERISTICS

The quarterly mean concentrations of selected water column characteristics are also presented in Table 3. These values are averages of data from four stations in the lake. While it is not readily apparent from the data presented, two features of the Long Lake water column are of particular significance. The lake does not undergo thermal stratification during the summer months and there is no extensive oxygen depletion in the bottom waters. Vertically the lake is fairly well mixed all year round.

A more detailed presentation of the water column data is given in Figures 5 and 6. These data are mean water column concentrations based upon samples collected at the four lake stations.

Primary production averaged 454 ± 121 mg C m^{-2} day^{-1} over the period. The growing season begins in March and extends through October (Figure 5). Primary production through the summer months was fairly comparable for both 1976 and 1977 the average values for the July to September period being 766 ± 151 and 837 ± 148 mg C m^{-2} day^{-1}, respectively.

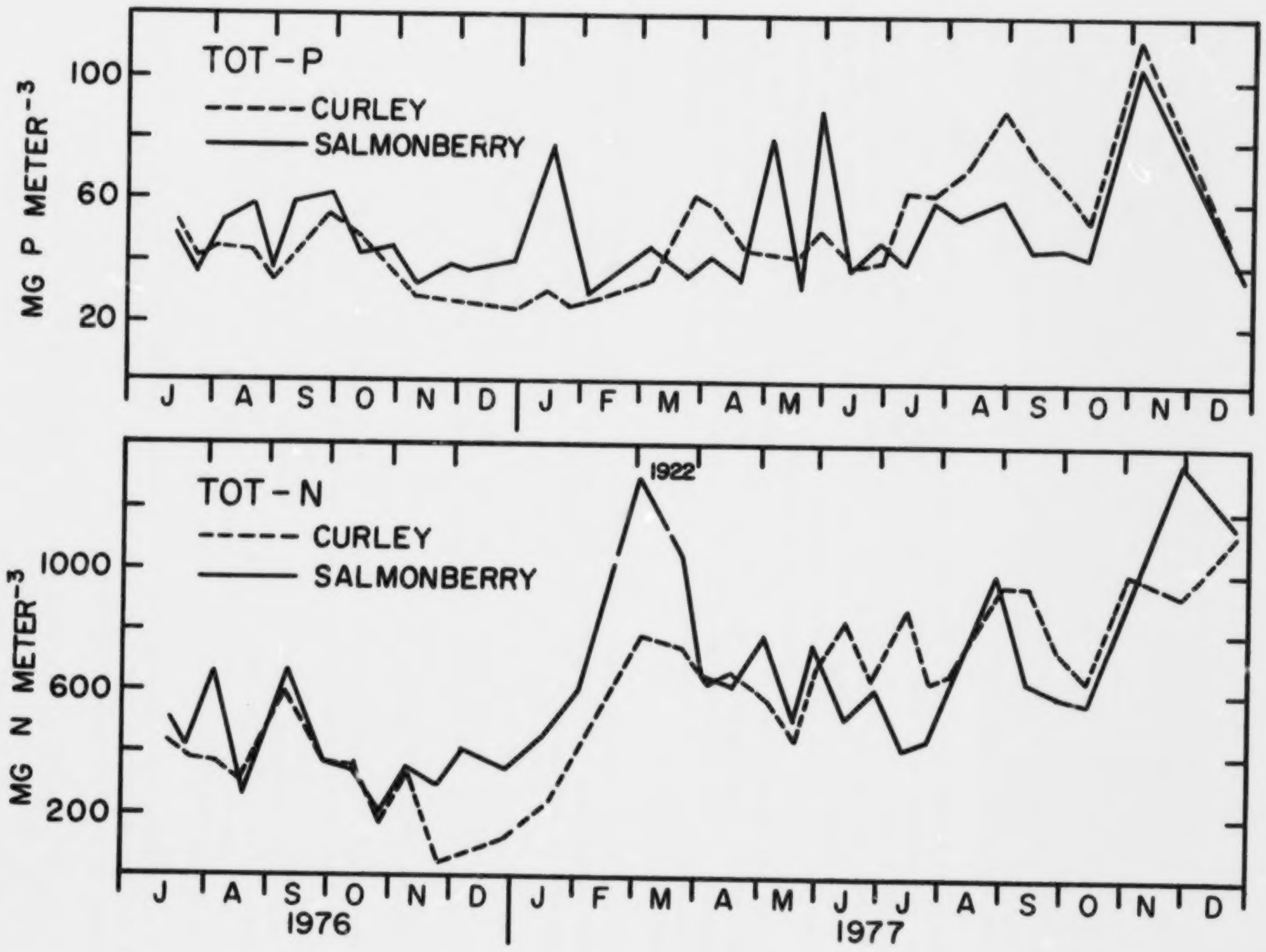


Figure 4. Seasonal variation in the concentration of total phosphorus and total nitrogen in Curley and Salmonberry Creeks.

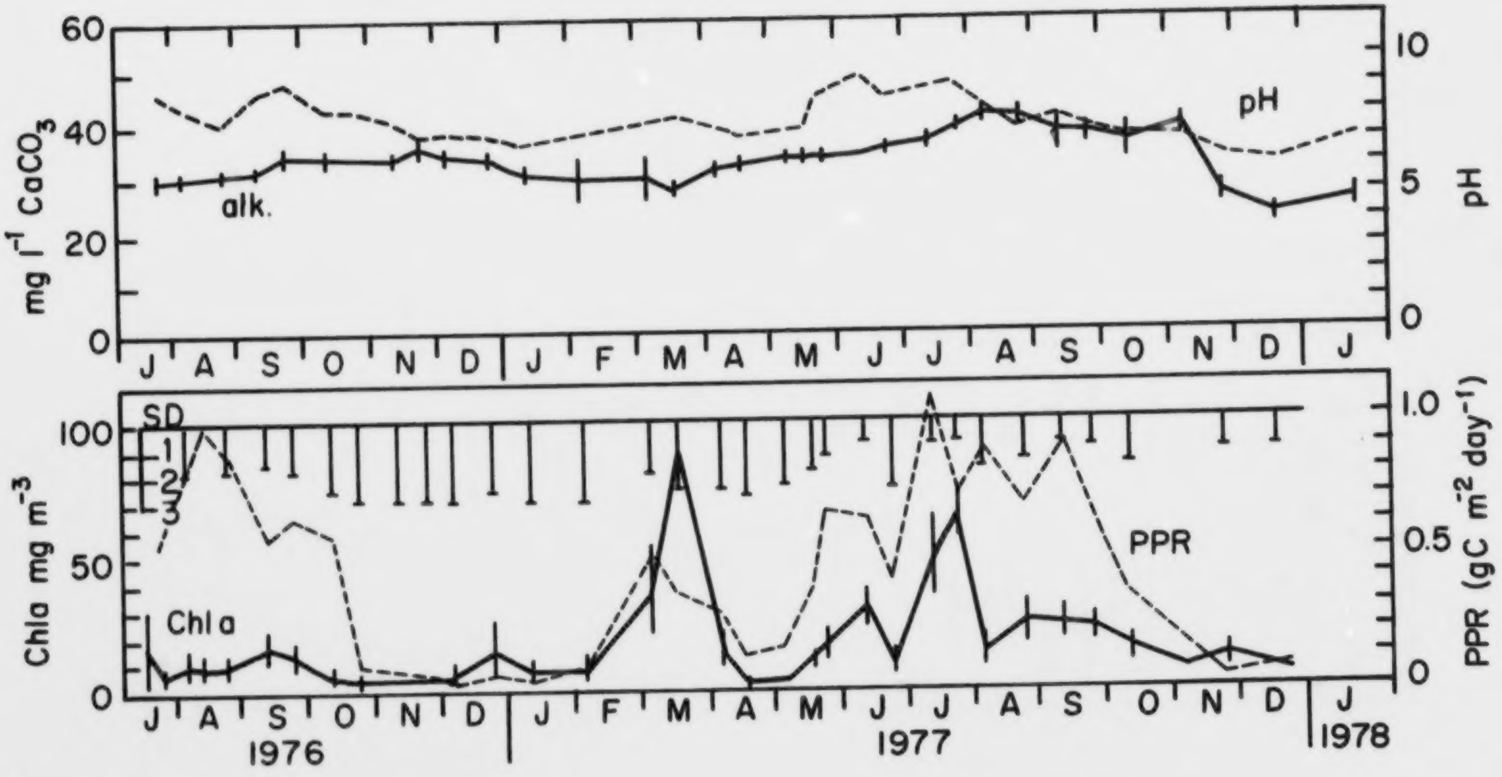


Figure 5. Biological characteristics of the Long Lake water column (means of 4 stations).

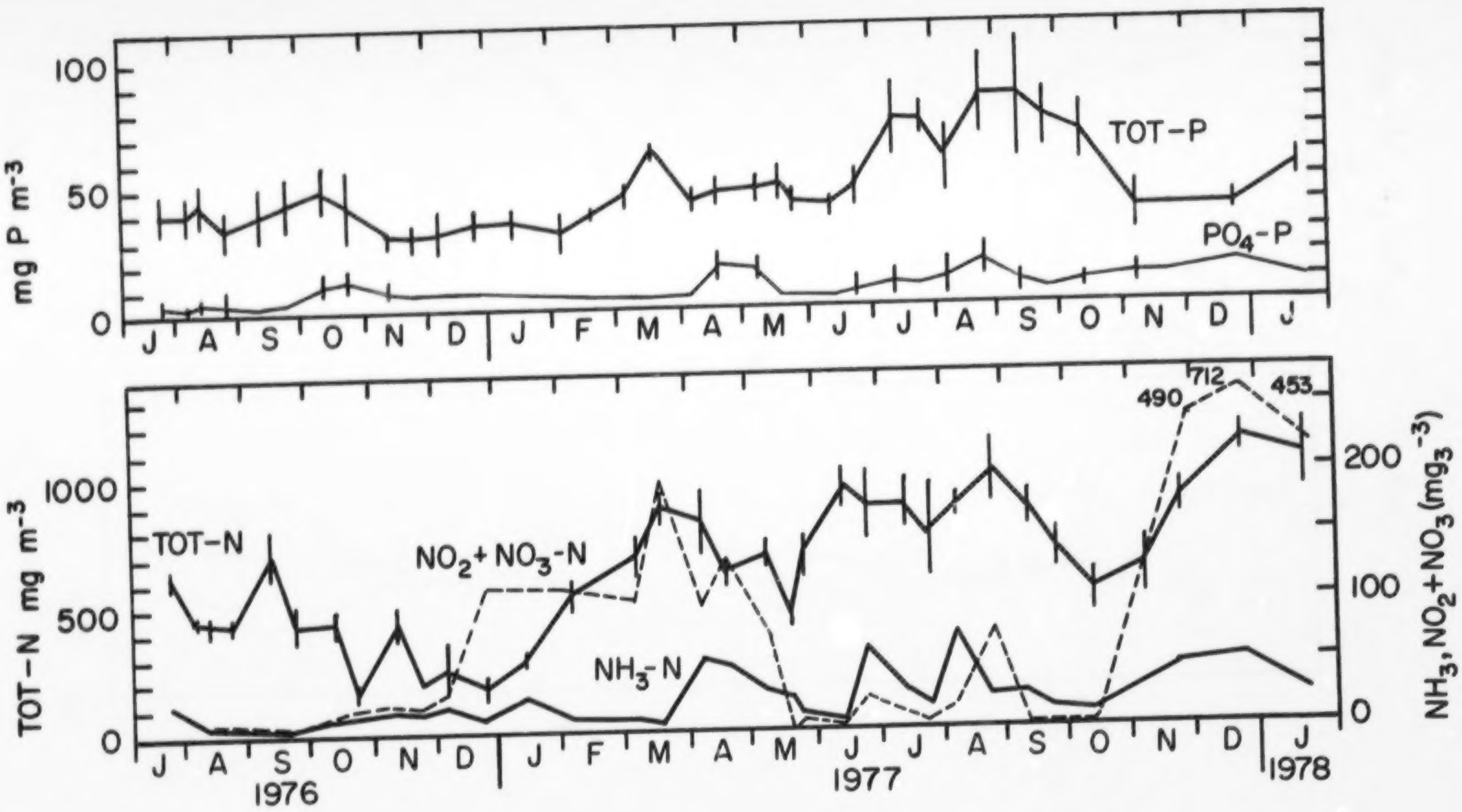


Figure 6. Inorganic nutrient characteristics of the Long Lake water column (means of 4 stations).

Chlorophyll *a* concentrations averaged 17.1 ± 5.1 mg m^{-3} with pronounced peaks occurring in March 1977 (83.4 mg m^{-3}) and July 1977 (60.5 mg m^{-3}). Chlorophyll *a* concentrations in the summer of 1977 (July to September) were considerably higher than those occurring for the same period of 1976, the average concentrations being 10.2 ± 3 in 1976 and 29.1 ± 9 in 1977. Secchi depth averaged 2 meters in July, 1977. Secchi depth closely followed chlorophyll *a* concentrations (Figure 5).

The increase in chlorophyll *a* during the summer of 1977, over that in 1976, may be related to an increase in inorganic nutrient concentrations beginning in January of 1977. The seasonal patterns of inorganic nutrients are shown in Figure 6. Clearly, the nutrient levels in 1977 were considerably higher than those in 1976. For the July to December period, the 1976 values for total phosphorus averaged 36 µg P liter^{-1} in 1977.

As can also be seen in Figure 5, Long Lake has a rather low alkalinity averaging around 30 mg l^{-1} and a pH usually between 7 and 8. However, pH exceeded 9.0 during both summers as a result of the high rate of photosynthesis. As a consequence of this increased pH, alkalinity subsequently increased approaching 40 mg l^{-1} in 1977.

An important observation with regard to the internal source of phosphorus in Long Lake can be illustrated from the seasonal distribution in the water column (Figure 7). The large water column concentrations of phosphorus during July-September 1977 appear to emanate from the bottom sediment. The higher concentrations at the bottom, which were in excess of 100 µg l^{-1}, contributed to keep the overall water column concentration near or above 80 µg l^{-1} for that three-month period when algal biomass was also greatest.

While much of the water column data are presented as averages for the three lake stations, north, mid and south, a considerable difference in biomass existed among the stations. In particular, nutrient content and plankton algal biomass were usually less at the southern most station and, at one, in an especially thick, lily pad dominated weed bed. Figure 8 shows the quarterly average values for total phosphorus and chlorophyll *a* at the four stations compared with an overall lake average (a cross bar). Note that the difference is most striking during July-September for both years and both constituents. This difference could be caused by any one or combination of three or more factors as follows:

1) a competitive advantage for nutrients, or through inhibition in favor of the higher density of macrophytes over plankton algae at the south station;

2) reduced turbulence at the south station because of denser macrophyte stands, creating a greater plankton loss rate by sedimentation compared to the north and mid lake;

3) the south lake portion is isolated from inflow during summer because the inflow is located opposite the midlake station (see Figure 1), which may tend to cause flow short-circuiting to the north.

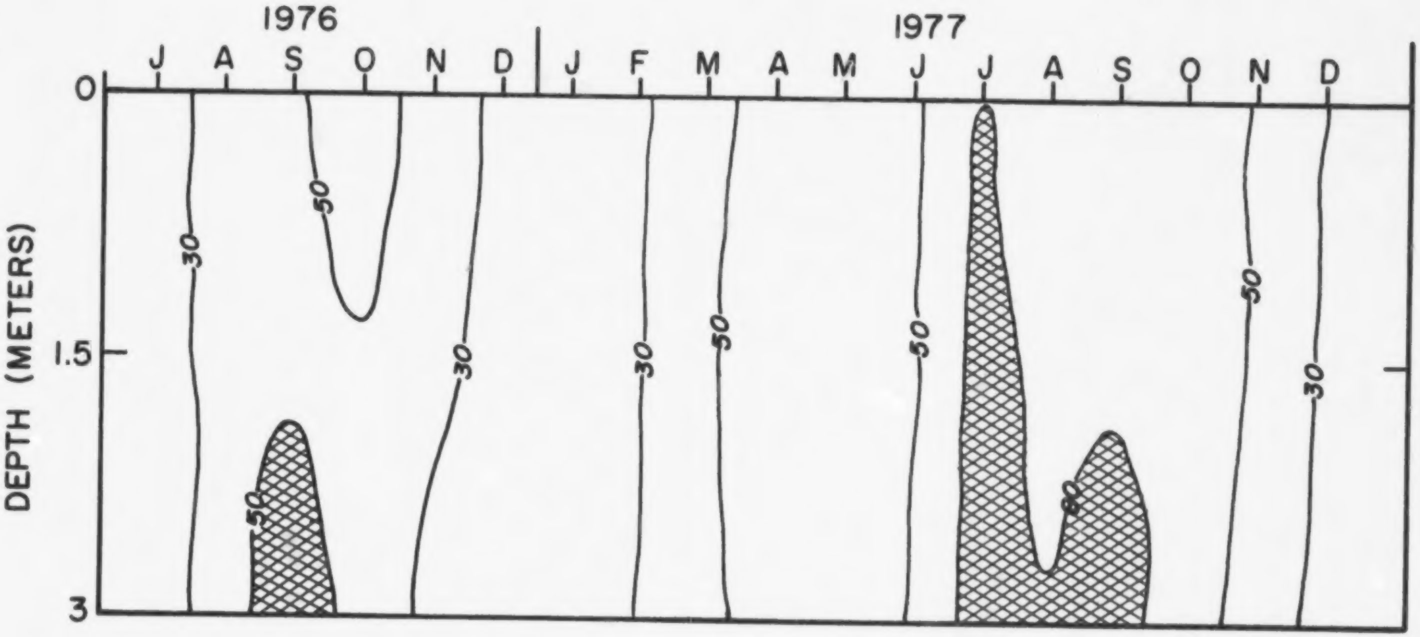


Figure 7. Total phosphorus isopleths for Long Lake for the period July, 1976 to December, 1977. Shaded areas represent periods of release from the sediments.

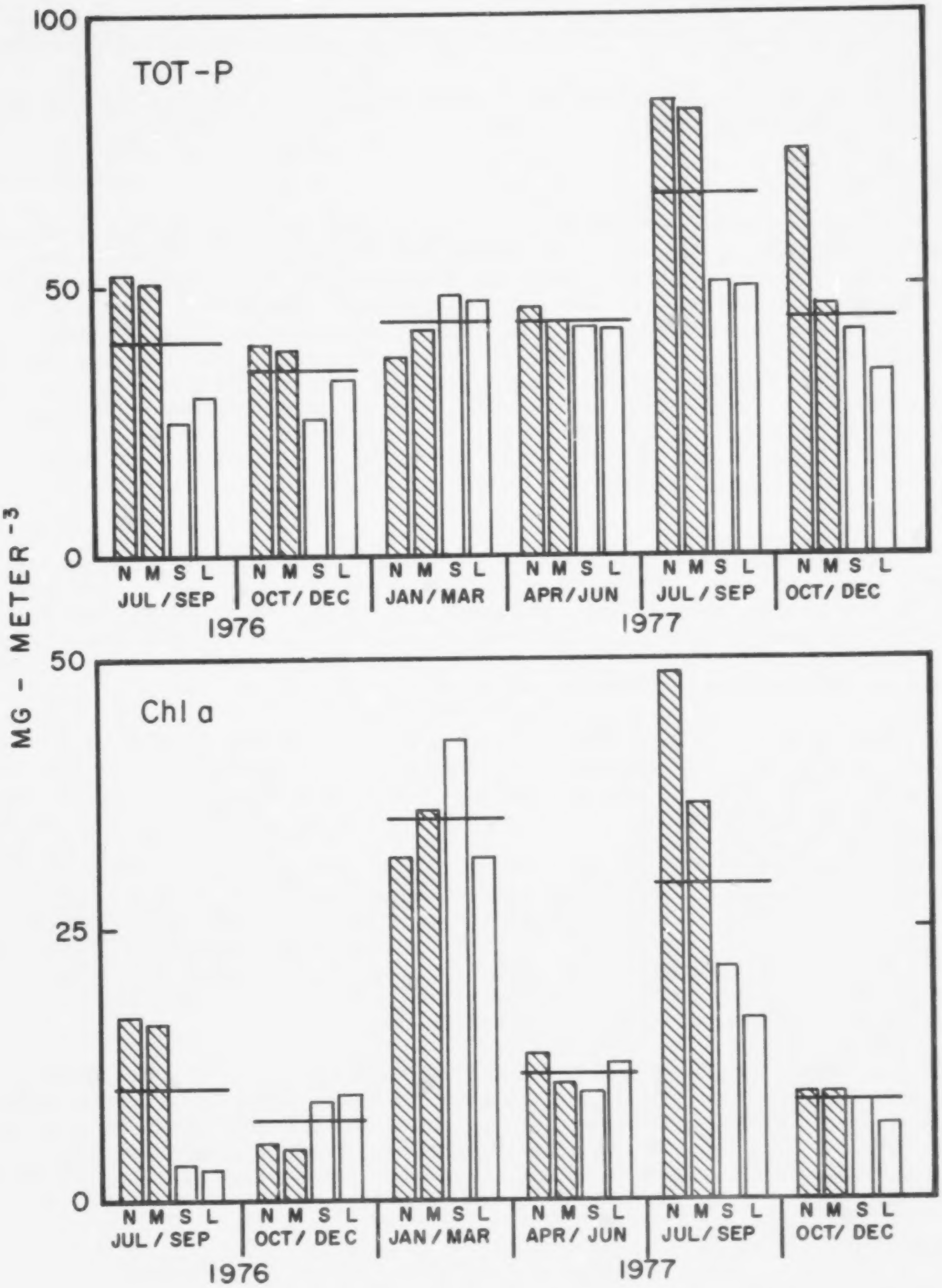


Figure 8. Variation in total phosphorus and chlorophyll a at the 4 lake stations.

The cause for this difference is presently under investigation but as yet no definitive information is available.

SEDIMENTATION

The measured flux of sediment in Long Lake is no doubt an overestimate of the gross downward rate of autochthonous and allochthonous sediment. The lake does not stratify, so traps cannot be placed below the thermocline and avoid the effect of summer turbulence and resuspension. The extent of this error is not known, except attempts are under way to determine the fraction resuspended. Nevertheless, the annual rate of this sedimentation for the lake was 1403 Kg P, or 1.03 g P m^{-2}, with the summer period showing the greatest magnitude. Very likely a large fraction of this trapped material may originate from macrophyte detritus and further it may be resuspended and resettled. For now, and for purposes of simplification, this rate of measured sedimentation, including resuspended sediment and macrophyte detritus, will be referred to as the gross rate.

The average net sedimentation rate, determined by analyzing two 30 cm cores from the center of the lake, was 415 g dry weight m^{-2} yr^{-1} in the recent sediments (since 1900). This amounted to 0.5 g P m^{-2} yr^{-1}. The data on stable lead indicated a total, permanent accumulation of 25 cm since 1900 with an early rate of about 0.32 cm yr^{-1} to about 0.43 cm yr^{-1} in recent years, which is rather typical of lakes in the area.

The technique of estimating sedimentation rate was that of stable lead using verification of aluminum and phosphorus to identify years of major flooding. The principal tag for stable lead in the area is the initiation of the internal combustion engine around 1925. This has been documented for Lake Washington by Schell and Barnes (1974) and reconfirmed in other lakes in the area by Spyridakis and Barnes (1977). There was yet another source during 1890-1913 and that is the American Smelting and Refining Company's smelter in Tacoma. The three periods of high sedimentation are indicated in Figure 9. Also indicated are the peaks in the aluminum/phosphorus ratio, which are indicative of associated floods (high Al, low P), and agree rather well with dates identified by lead. The sedimentation rate was also verified with Cs^{137} which was deposited from bomb blasts during 1955-63.

The sedimentation rate estimated from sediment cores is a net rate. That is, the 0.5 g m^{-2} yr^{-1} thus includes either a large fraction of resuspended or internally released and sedimented phosphorus and the difference between these two rates, 0.52 g m^{-2} yr^{-1} can for the present at least, represent an estimate of the internal loading from plants, sediment release and particulate resuspension.

MACROPHYTES

The macrophytes in Long Lake are dominated by _Elodea densa_, a larger and much leafier species than _E. canadensis_, at least in this lake. While _E. canadensis_ was present, it comprised only a small percent of the biomass. _Potamogeton praelongus_ was rather abundant in the south end during the surveys although subsequent observations have shown it to be most abundant in the

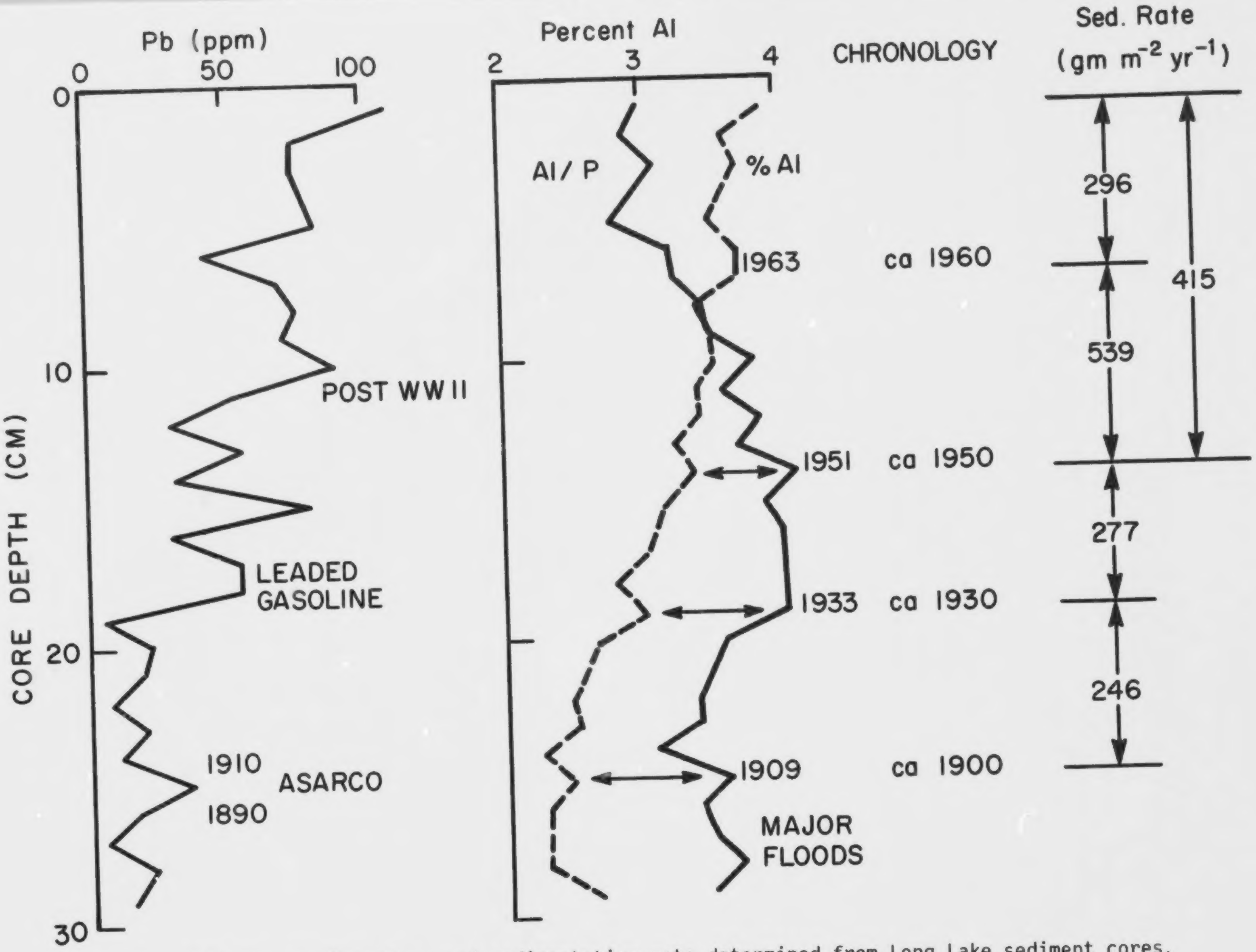


Figure 9. Sediment geochronology and sedimentation rate determined from Long Lake sediment cores.

spring and dying back somewhat by autumn when the surveys were performed. Other species present are Traphar, Brassenia, and Ceratophyllum.

Aerial photographs show the south end of the lake to be most populous with macrophytes. While this is true nearly all of the lake's bottom is inhabited by E. densa. Although it grows to a height of 6 meters in the deepest areas, its nearly complete coverage isn't generally recognized since it does not reach the surface.

The results of the two surveys are shown in Table 4. Although the mean biomass was slightly greater in 1976 than 1977, the survey was one month later, which probably allowed for some break up and decomposition to occur in 1977. Nevertheless, the larger algal crop in 1977 than in 1976 no doubt had a significant influence in reducing the light penetration and growth of macrophytes.

TABLE 4. MACROPHYTE BIOMASS (GRAMS DRY WEIGHT METER $^{-2}$) IN LONG LAKE.

DATE	LOCATION	% AREA	species	n	$\bar{x}$ (gm/m^2)	s	cv	LAKE MEAN
Sept. 1976	North	11.0	c,d	18	150	119	79	259 ± 50
	Mid	59.1	c,d,0	18	238	147	62	
	South	29.9	d,n	8	341	130	38	
Oct. 1977	North	11.0	c,d,p	8	138	65	47	192 ± 53
	Mid	59.1	d,p	9	112	97	87	
	South	29.9	c,d	6	371	150	40	

Species: c = E. canadensis, d = E. densa, p = Potamogeton, n = Nuphar

Lake means and confidence intervals based upon area weighted means and variances where:

$$CI = 2\sqrt{\hat{V}_{\bar{x}}}$$

The rather even coverage of the lake does not require too large a sample size to insure a reliable estimate of the mean. Note the relatively small confidence interval considering the usually difficult spatial problem that macrophytes often present.

An important role of the macrophytes in restoration effectiveness may be in their contribution to the phosphorus budget. The percent P of the plant dry weight averaged 0.3 ± .11. Thus, average biomass in 1976 and 1977 would have represented a mass of P in the lake of 1,068 and 795 Kg, respectively.

As will be seen in the next section about 85 percent of that P was probably mined from the permanent sediments through the plant roots unless some refractory fraction could be part of the internal P input to the lake.

DISCUSSION

Long Lake is highly eutrophic and shows effects from dense blooms of blue-green algae during the entire summer as well as from a dense stand of macrophytes, principally Elodea densa, that occupies nearly the entire lake. Largemouth bass and black crappie are the most abundant fish in the lake and have an estimated density of 42 and 102 fish per hectare, respectively (105 and 255 per acre); not a particularly dense population (Congleton, personal communication). Few bass are in excess of 25 cm length.

In order to restore this lake to some less objectionable status, it is necessary to know the source of nutrients, particularly phosphorus, that is responsible for the abundance of plant material and the resulting degraded quality. The first point that becomes clear is that its eutrophic state is probably not caused by the external nutrient loading. The external load of P is 390 kg yr^{-1} or 0.28 g m^{-2} yr^{-1}. The calculated critical loading from Vollenweider (1976) is 0.53 gm^{-2} yr^{-1} according to:

$$L_c = 200\ (\bar{z}\rho)^{0.5}$$

where L_c is the critical loading, $\bar{z}$ is mean depth and ρ is flushing rate. Clearly, Long Lake should not be eutrophic if its principal P loading is from external sources. While 0.28 g m^{-2} yr^{-1} is a considerable quantity of P for a shallow lake, the high flushing rate insures that much of that P will be washed out of the system before it can be used.

Further, a prediction of Chl a from P loading (L_ρ) according to Vollenweider (1976):

$$\text{Chl } \underline{a} = 0.376 \left[\frac{L_\rho}{qs(1 + \frac{\bar{z}}{qs})} \right]^{0.91}$$

where L_P is aeral P loading and qs is surface hydraulic loading (m yr-1), gives only 7.8 μgl^{-1}. The observed average concentrations for Long Lake were greater, 10 and 29 μgl^{-1} for the two summers. Thus, one must conclude that Long Lake does not behave in the same way as most lakes studied, with respect to phosphorus loading. The reason is probably that there have been no or few unstratified shallow lakes in the data sets analyzed by Vollenweider, Rast and Lee, (1978) and others for the relation between trophic state and external loading. The logical additional source of P that could explain the eutrophic state of Long Lake is probably internal.

The phosphorus budget has several uncertainities but for the most part is rather accurate and is shown in Table 5. The principal difficulty is that it is based on a very dry year. The interesting point is that the outflow nearly matches the inflow, 384 versus 390 Kg yr^{-1}, and there is a net increase of P in the lake water of 89 Kg yr^{-1}. This is highly unusual inasmuch as most lakes discharge only on the order of 35 to 40 percent of the entering P, the remainder being deposited in the sediments. Long Lake actually discharges more P than comes in during some quarters. At first glance, this implies that a sizable internal source exists, assuming that the external sources are reasonably accurate, and that seems reasonable in view of the fact that the water budget balanced reasonably well.

As Table 5 shows, the internal source can be estimated by difference if a reasonably good measure of sedimentation rate is available. The gross rate of 1,403 Kg yr^{-1} no doubt results in an overestimate of the internal source, 1486 Kg yr^{-1}, because of all the resuspended sediment and plant detritus that is included in that measured rate. The equation (Table 5) is actually more appropriate for use with a net sedimentation rate, in other words a permanent annual burial of P, and that is obtainable from the core analysis. Using 682 Kg yr^{-1} gives an internal loading of 765 Kg yr^{-1}, which is the most reasonable estimate of internal loading. This still may be too high because no correction was made for the preferential deep water deposit of particulate matter.

TABLE 5. LONG LAKE TOTAL PHOSPHORUS BALANCE 10/76 - 9/77

	INFLOW (KgP)			LOSSES (KgP)		$\Delta\Sigma P$	P_{int}
Period	Si	GW*	Pre	So	Sed	(KgP)	(KgP)
10/76 - 12/76	49.50	2.86	3.53	51.81	355	0	351
1/77 - 3/77	153.20	0.06	8.26	151.86	223	+26.8	240
4/77 - 6/77	105.57	2.83	4.12	93.92	273	- 2.7	252
7/77 - 9/77	49.24	5.72	5.12	86.73	552	+64.56	643
Σ	357.51	11.47	21.03	384.32	1403 (682)**	+88.66	1486 (765)**

Si = Surface inflow; GW = ground water; Pre = precipitation; So = Surface outflow; Sed = Sedimentation; $\Delta\Sigma P$ = change in lake concentration; P_{int} = calculated internal source.

$P_{int} = So + Sed + \Delta\Sigma P - SI - Pre - GW$.

* ground water inputs calculated on ave. P concentration of 72 mg m^{-3}.

** calculations based on sediment core analysis giving sedimentation rate of 415 g dry wt m^{-2} yr^{-1} with an average P concentration of 0.12%.

The specific source of the internal load is unknown for certain. Laboratory P^{32} experiments suggest that plant excretion is not the source, since the experimental plants showed no net loss to the water, even though 85% of their P was taken from the sediments. However, plant biomass in the fall of 1976 contained 1,060 Kg of P in the tissues. While plants do not completely die back in winter, the biomass is nonetheless greatly reduced. Also, part of the P in the biomass must be refractory. However, even if one half of their P had been released upon decomposition in the winter of 1976-77 it could have contributed a sizable portion of the 765 Kg.

In addition, there are the processes of resuspended particulate matter and release of dissolved P from interstitial water. While the latter is not known to be extensive under aerobic conditions, Figure 7 nonetheless indicates that some increase did occur, but whether primarily a biological or chemical process is not known. Thus, the internal source is no doubt a combination of these three processes and possibly even including excretion as a fourth process, as the macrophytes age. The laboratory experiments were performed with young, vigorous plants that may not normally be prone to excretion.

If the internal source is coming from sediments, and it is nearly equal to the quantity being permanently buried each year, then there must be some redistribution of P from shallow areas to deeper areas. This is not unreasonable, however, as such transport outward conforms to normal processes in lakes. What it could mean, however, is that the sedimentation rate, based on mid-lake cores, is overestimated if applied to the whole lake for the reason just given. If so, then the rate should be corrected to more of an average for the lake, which would serve to lower the internal source.

The surrounding houses (121) cannot be entirely disregarded as an additional external source, except it was felt that the half barrels should reflect such inputs. The barrels were placed proximal to shoreline houses in hopes of spotting larger concentrations, but none developed. In any event, the source from septic tanks would not be larger than 100-200 Kg yr^{-1} and probably less.

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THE MONITORING OF RESTORATION EFFORTS AT COLLINS LAKE, VILLAGE OF SCOTIA, NEW YORK

by

C. J. George*, P. L. Tobiessen*, P. D. Snow**

FORMALITIES

On January 8, 1976, the New York State Department of Environmental Conservation in cooperation with the Village of Scotia, Schenectady County, New York, was awarded a matching grant (S804250010) in the amount of $46,250 for dredging and other restorative activities for Collins Lake in the Village. This award was made under the provisions of PL-92-500/Section 104 as administered by the Environmental Protection Agency. Subsequently, the reporting investigators of Union College, Schenectady, New York, were awarded a grant (R804572010) on 22 July, 1976, under the same granting provisions for the purpose of monitoring the restoration efforts at Collins Lake. On October 10, 1976, the Village was granted permits (No. 447-04-007 and 447-76-126) for dredging in accord with Article 24 (Freshwater Wetlands) of the New York State Environmental Conservation Law including stipulations that dredging not exceed 8 feet in any place, that areas in the eastern part of the lake less than 1 m depth not be dredged and that emergent vegetation at the outlet not be disturbed.

On 15 December, 1976, the Village was assigned a work permit (No. 9953), following standard public notification (No. 8643, 17 September, 1976) by the New York District of the Corps of Engineers under provisions of the several relevant federal laws. The dredging contract was finalized in the Spring of 1977 by the Village in concert with regional offices of the EPA and the New York State Department of Conservation.

THE LAKE AND ITS PROBLEMS

Collins Lake is an oxbow lake derived from a northward meander of the Mohawk River. The basin was initially isolated along its southern aspect by naturally placed river sediments but this barrier has been raised further by the building of a dike in 1804 which was enhanced as a carriage route in 1805. Later, especially in the 1940's, it was raised still further through the deposition of many thousands of cubic meters of diverse fill and river dredgings. The resulting barrier has greatly reduced flooding, with the incidence of only one earlier flood, and one this year in March, contributing in each case about 1 m of turbid water to the lake. In the flood of 1976, one sector of the dike was eroded away resulting in the dispersal of about 75 m^3 of ashes over the southern aspect of the lake shore and into the lake proper.

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The eastern end of the lake has also been closed by earthen fill. A map of 1799 by Claude Joseph Southier prominently shows a road to the east of the lake in approximately the position of the existing causeway, a feature which must have increased the lake area to some extent. In 1805 a bridge was built across the river and linked to the same causeway. At this time, the outlet of the lake was restricted to a passage between bridge abutments, raising the lake some additional amount. In 1945 or 1946 the bridge was replaced with a culvert the river side of which was equipped with a flapper valve designed to prevent the movement of river flood water into the lake.

The new installation resulted in the raising of the lake to its current approximate level of 216' and an aerial enlargement to its current extent 22 ha (55a). The joint action of the southern or Schonowee dike and the Washington Avenue causeway has thus been to isolate the lake from the river and to accent the influence of the various springs located at the foot of a major sand aquifer on the west and northern edge of the lake. These springs run actively year around maintaining circular openings in the ice and a zone of unfrozen water along the northern and western shores. Our divers have inspected one of these springs in early March noting at a depth of 3 m an open tube in the bottom about 15 by 5 cm in extent surrounded by a circular "sand boil" area about 4 m in diameter. An abundant spring flow appears to thus constitute a major portion of the water entering the lake.

The lake has long been a recreational asset to the village and region. In the 1950's, Collins Park, located between the river and the southern shore of the lake, was enlarged and a swimming area was developed along the central part of the southern shore. A thousand or more cubic meters of sand were introduced to form a sand beach. Concurrently, an adjacent storm sewer was closed and another opened immediately north of the outlet at the eastern edge of the lake.

It is reported that the river sewer may become contaminated with household sewage at time of heavy storm runoff. The swimming facilities of the lake are intensely used during the summer months and this may result in a significant contribution of organic nitrogen to the system. Other sources of plant nutrients and pollutants have been runoff contaminated by snow and leaves dumped at lakeside. These practices have continued through 1976 but the spoils area with its dike is designed to contain snow melt waters and leaf breakdown products toward abatement of the problem.

The augmented nutrient supply and isolation from the scouring influence of the river appears to have favored the establishment and more troublesome proliferation of the water chestnut, Trapa natans, and the curly leaved pond weed, Potamogeton crispus. The water chestnut emerged as a major pest in the early 1900's and spread into the Mohawk River requiring much expensive control effort. Today the species still survives in the lake, some seven bushels of nuts being removed during the summer of 1976 and about 4 during 1977. The curly leaved pond weed is currently the most conspicuous and detrimental.

RESTORATION METHOD

The major manifest problem at Collins Lake is thus viewed as excessive growth of the curly leaved pond weed, *Potamogeton crispus* during the spring and early summer. Associated problems are shoaling due to the accumulation of organic matter and the development of anoxic deeper waters. The main causes of these problems are believed to be the inevitable processes associated with lake aging as accelerated by increased influx of phosphorus and the introduction of exotic plant species such as the forenamed pond weed and (earlier) the water chestnut, *Trapa natans*. The main planned attack on the problem is to reduce the input of phosphorus to the lake through the stopping of the lake-side dumping of leaves, other organic matter, and snow; to improve maintenance of a flapper valve at the outlet designed to exclude nutrient rich flood waters; and to remove from the lake proper about 100,000 m^3 of the accumulated organic matter with its associated plant nutrients which are continually (we suspect) being recycled by the pondweed. The means of removal has been a hydraulic dredge developed by Mud-Cat division, National Car Rental, Inc. The organic matter is aspirated from the bottom, causing little turbidity, and pumped to a decanting lagoon situated at the southeast edge of the lake and the supernatant water is returned to the lake. Details on this process are presented in later paragraphs.

The storm water outfall located at the eastern edge of the lake is near the outlet and thus much of its water is immediately discharged from the lake but during periods when the Mohawk River exceeds the lake level, i.e. 216' a.s.f., the flapper valve is forced closed and storm waters enter the lake. Rather than shifting the problem to the river by relocating the outfall, a berm well populated with aquatic plants is to be developed surrounding the outfall and outlet, thus limiting the impact of storm water on the main body of the lake.

Dredging commenced in July of 1977, and lake-side dumping has been discontinued and an improved maintenance program for the flapper valve at the outlet has been instituted.

The decanting lagoon with an area of 2.4 ha (6a) and an average holding depth of 2 m functioned well with much of the initial water passing into the ground or through the porous matter of the dike before it reached the sill level of the outfall pipe. Water leaving the lagoon entered a swamp-marsh area with nutrient and solid concentrations less than those of the water stream leaving the marshland and entering the lake.

Roughly 30,000 m^3 (bathymetric basis) have been removed thus far. This volume has effectively reduced the volume of the lagoon by roughly 60%. Some dewatering of the in-place sediment is expected under the influence of freezing and thawing and thus a renewal of capacity; however, if this does not occur sediment will have to be removed, additional decanting space must be found, or the project will have to pause until warm weather dehydration enlarges the storage prism.

Total solids entering the lagoon from the dredge vary from 35 to 55 g/l. At the outlet, after a theoretical settling time of 3 days, the suspended solids concentration is between 25 and 50 mg/l. This yields an effective removal of 99.9%. Effluent values for nutrients were: 50 µg/l-total P, 25 µg/l-ortho P, 0.16 mg/l NH_3-N, and 0.8 mg/l NO_3-N.

A public relations program centered on news releases and public lectures has informed the public of intentions and progress and excellent public rapport has been maintained. Dredging proceeded concurrently with swimming, boating and fishing without any detected negative response. Numerous fruits of the water chestnut were floating during the dredging process but a prevailing southwesterly wind kept them away from the swimming beach during the swimming season. Odors, sounds and turbidity associated with dredging were negligible and caused no public commentary or criticism.

MONITORING TARGETS

Four stations have been established and are visited fortnitely for the sampling at several depths of water for the evaluation of physical, chemical and biological parameters, i.e. O_2, soluble orthophosphate, total phosphorus, NO_3, NH_3, alkalinity acidity, pH, hardness, conductivity, T, Secchi disc depth, numbers and kinds of phytoplankton and zooplankton and concentration of chlorophyll. Heavy metals are also being surveyed in cooperation with the New York Department of Health. Concurrent gill netting at one site is directed toward the capture of golden shiners, Notemigonus crysoleucas, and yellow perch, Perca flavescens for routine morphometry and histology of the liver, spleen, kidney and gonads. Two transects are examined quantitatively for aquatic macrophytes with primary attention being given to the numbers and biomass per square meter of the curly-leaved pondweed, Potamogeton crispus.

Numbers which are produced are applied to computer cards for storage, analysis and graphic print-out as demonstrated later in this report.

MONITORING RATIONALE

The objectives of the dredging and improved maintenance program have already been stated and focus on reducing weed growth, increasing lake depth and improving aeration of the deeper waters while at the same time not causing untoward and long-enduring consequences. Our monitoring program from the onset has thus included the macrophyte assay, bathimetry and routine oxygen studies, and in that phosphorus is thought to be the key limiting nutrient, evaluation of this parameter has been given special attention. Toward monitoring for untoward consequences we have followed a baseline approach whereby various parameters are defined for about 1 year before dredging with a fervent hope that other major variables, more impactful than dredging, do not arise and dominate the situation. Unfortunately, the floods and heavy snows experienced during the last few months of the study may be influences of this very significant kind. We remain hopeful, however, that we will be able to sort out the influences.

Within the monitoring program several questions have emerged as especially relevant. The first is the matter of interaction between planktonic and rooted primary producers. Thus far we sense that rooted macrophytes such as Potamogeton crispus, which are able to grow at reduced light intensities and therefore at greater depths, may play an important role in the regeneration of key plant nutrients and their release to the water column. At the same time, they may effectively remove key nutrients from their ambient waters, thus suppressing planktonic primary producers. The death and breakdown of these plants, however, may foster a dramatic resurgence of planktonic growth which might otherwise have been impossible. There is the possibility that dredging to depths greater than those tolerated by P. crispus may greatly reduce nutrient regeneration, reduce the primary productivity of rooted forms and direct nutrients into the phytoplankton, which in turn would be swept from the lake by spring waters with low nutrient concentrations.

If indeed the production of oxygen demanding organic matter and anaerobic water can indeed be reduced, the nutrient regeneration occurring in the deeper, western basin of the lake may further reduce the eutrophy of the system while at the same time increasing the living space for benthic invertebrates, fish and, with time, perennial macrophytes, which are nutrient conserving.

PROPERTIES OF THE LAKE

VOLUME OF DISCHARGE

The averaged outflow from the lake as measured for the period 6/22/76 to 6/8/77 was 2.27 cfs with a range of 1.96 to 4.13 cfs (D. Howie). Because of the removal of the outlet on June 8, 1977, as associated with the lowering of the lake for dredging, we have not maintained a record of flow volumes.

OXYGEN AND TEMPERATURE

The variations in O_2 and temperature (Figures 1 and 2)* are typical for a northern lake with moderately high biological activity. Graph 3 (percent saturation) yields the best interpretation of O_2-Temperature variation as a function of biological activity. During the fall of 1976 and 1977, values below saturation are attributed to bacterial degradation of dead plant and algal matter. Ice cover in 1976-1977 reduced the amount of dissolved oxygen due to the absence of atmospheric transfer of O_2, absence of light for photosynthesis, and bacterial breakdown of residual organics in the water column.

The two zones of supersaturation have different origins. The first, from March to July 1 corresponded to ice melting, river water input and especially the tremendous growth of the macrophyte, P. crispus. Increases were also apparent in phytoplankton, but had a minimal effect. Death of P. crispus after July 1 and decay of this plant matter is believed to be the major cause of undersaturation. Subsequent to the death of P. crispus, the release of nutrients back into the water column, and removal of macrophyte competition, one observes the later summer algal bloom and supersaturation during this time.

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* All figures and tables are included at end of text.

TEMPERATURE, OXYGEN AND PERCENT SATURATION VERSUS DEPTH

Figures 4 and 5 illustrate a typical stratification occurring in August, 1977 with anaerobic conditions below 5 meters. The percent saturation versus depth plot in Figure 6 is most indicative of algal supersaturation in the top 2 meters to anaerobic conditions below the thermo-chemocline. Decay of suspended and benthic organics by bacteria plus the obvious lack of vertical mixing is the main cause of hypolimnetic depletion of O_2.

PHOSPHORUS VERSUS TIME

Variations of total phosphorus in (μg/l as P) versus time are shown in Figures 15 (surface) and 16 (8 meters) for the west station. Surface concentrations of total P remain fairly low (20-25 μg/l) during the winter. Spring overturn and river in-flow increased the concentrations to almost 60 μg/l in the spring. The constant decrease (until July 1) is probably due to P. crispus uptake and perhaps co-precipitation of phosphate with $CaCO_3$. The rapid increase in July is mainly due to release of ortho and organic-P from bacterial breakdown of dead P. crispus (similar to ammonia increase). Also, because of temperature, pH, and redox potential changes, a substantial amount of orthophosphate may have been released from the sediment in the shallow parts of the lake. The late summer decrease is attributed to algal uptake of orthophosphate whereas the fall increase was probably from the combined breakdown of dead algae and lake overturn (see Figure 16).

Anaerobic (reducing) conditions in the hypolimnion and stratification during the summer are the major reasons for the extremely high (550 μg/l) concentrations of phosphorus shown in Figure 16. Ferric phosphate and allied ferric hydroxy phosphate compounds appear to limit the amount of ionic phosphate in the bottom (8 meters) water when the system is oxidizing. During the summer, phosphorus associated with ferric complexes is released due to the reduction of ferric iron to ferrous iron. Interstitial phosphate can therefore flux out of the sediment and concentrate in the hypolimnion. A rapid decrease is noted after overturn due to dilution with surface water and chemical precipitation of orthophosphate in ferric compounds. Interestingly, comparisions of surface and bottom water concentrations of phosphate yield, during the summer, a ten-fold difference. Surface values were about 50 μg/l whereas bottom values were about 500 μg/l. Release from the sediment interstitial water under anaerobic conditions appears to be responsible for the tremendous gradient.

Figure 17 shows the correlation of total phosphorus and orthophosphate in the bottom (8 meter) waters. Low values are from aerobic conditions where about one-half of the total is orthophosphate. Under anaerobic conditions, almost all (97%) of the total phosphate is orthophosphate. This again indicates release from the sediment of orthophosphate from the interstitial water and the breakdown of ferric phosphate compounds which would yield orthophosphate. Little, if any, of the total phosphate is associated with organically bound phosphorus.

Other interrelationships of the chemical-physical-biological systems within the lake are briefly described in Appendix A. Future analyses of the data will hopefully show more precise interrelationships.

NITROGEN - AMMONIA AND NITRATE VERSUS TIME

The temporal variations in NO_3-N (mg/l) shown in Figure 12; S for west station at 0.5 meters from September 14, 1976 to October 12, 1977. A late fall value is extrapolated on the curve to show increases in nitrate throughout the fall and winter as organic-N was oxidized to ammonia and to nitrate by bacteria. The extreme high in March is due to input of river water. The rapid decrease in early spring is probably due to P. crispus uptake being greater than the rate of nitrate evolution from ammonia. The small rise in June may be due to a decrease in the uptake rate of nitrate by P. crispus as they stop growing. Here evolution of nitrate from ammonia is greater than plant uptake. The well defined reduction of nitrate in the late summer is undoubtedly due to algal uptake. The fall increase, mirrored by an increase in ammonia, is due to a decrease in algal uptake and greater evolution of ammonia from the bacterial breakdown of dead plant and algal organic nitrogen.

Figure 13 of ammonia (NH_3-N, mg/l) somewhat follows the trends in nitrate. Late fall evolution of ammonia from organic nitrogen due to algae and plants is followed by a winter decrease in ammonia as more ammonia is converted to nitrate, i.e. rate of organic-N to ammonia conversion decreases as organic-N is depleted. A similar increase in ammonia is noted in March as 1 meter of river water floods the lake. This occurs at the same time as the spring overturn thus bringing high amounts of ammonia to the surface. The decrease of ammonia throughout the spring was due to its oxidation to nitrate and subsequent uptake by the plants.

An abrupt and rapid increase in ammonia is noted in early June due to the death of P. crispus and rapid breakdown of their organic nitrogen to ammonia. Also, the decrease in early fall is due to a lack of readily available organic nitrogen. As soon as algal growth decreases and bacterial breakdown of algae occur, the ammonia concentrations increase throughout the late fall. Interestingly, the concentrations of ammonia and nitrate are both 1 mg/l at the beginning of the spring and both vary, within limits, depending on algal and plant growth or death.

Figure 14 depicts change in ammonia in the bottom water (8 meters) versus time at west station. Low (1-2 mg/l) values occur during winter and early spring with aerobic waters and conversion to nitrates. When the hypolimnion becomes anaerobic, conversion of ammonia to nitrate ceases and high (5-7 mg/l) concentrations occur. Fall overturn, mixing, and aerobic conditions again decrease the ammonia that was evolved from the breakdown of organic nitrogen in the bottom sediments.

CALCIUM, MAGNESIUM AND ALKALINITY

The variations in total hardness (mg/l as $CaCO_3$) and alkalinity (mg/l as $CaCO_3$) versus time for west station at 0.5 meters are not shown in Figures 10 and 11. Interpretations of the data are mainly based on the equilibria of

$CaCO_3$. Dissolution or precipitation of $CaCO_3$ varies with pH, temperature, Ca, HCO_3, CO_3^-, CO_2, and photosynthetic changes of pH and alkalinity. The overall trend is the increase of calcium and bicarbonate during the winter due to lower pH, higher CO_2, and lower temperatures. $CaCO_3$ in the water column and sediment tend to dissolve. The rapid drop of values in the spring is due to increase in temperature, increase in pH, removal of CO_2 by macrophytes, all of these processes tending to cause $CaCO_3$ to precipitate. This was evident for the upper surfaces of leaves of P. crispus which were encrusted with $CaCO_3$ by June.

The death and decomposition of P. crispus in July appears to have caused the precipitated $CaCO_3$ to redissolve. The evolution of CO_2 and pH drop during decomposition appear to cause this change. The decrease of calcium in late summer is attributed to deposition of $CaCO_3$ due to a pH increase (and temperature increase) from algal photosynthesis. Additional alkalinity, greater than the amount released from $CaCO_3$ dissolution, were attributed to algal photosynthesis. This alkalinity also decreased in later summer due to $CaCO_3$ precipitation.

Winter variations of the three forementioned parameters and the program (TDOX) are shown in Table 2. Figures 7 and 8 illustrate the typical ice covered lake with slight density stratification in temperature and decreasing concentrations of O_2 below 5 meters. Percent saturation (Figure 9) indicates the uptake of O_2 by benthic bacteria and other organisms. During the months of February and March the depletion of O_2 (see Figure 3) continued and then the ice melted and lake overturn occurred. Overturn was apparently enhanced by river water (1 meter depth) entering the lake in March. Heavy metals have been examined for us by the New York State Department of Health in cooperation with Dr. Wolfgang Fuhs. Lake water, interstitial water of the sediments, ice and dumped snow have been examined with the highest levels appearing in the dumped snow. Lead concentrations were 3.9 and 1.5 mg/l of the resulting melt waters. In contrast, interstitial levels were less than 0.010 mg/l. Iron was also high in dumped snow with concentration of 15 and 5.7 mg/l of the melt water but interstitial levels were also high being 15, 9.7 and 3.7 mg/l for stations NE and N respectively. Copper was also relatively high in dumped snow with concentrations of 0.21 and 0.13 mg/l. Interstitial water concentrations were 0.05 and 0.06 for stations NE and N respectively. Additional analyses using X-ray fluorescence have also been performed.

Several chlorinated hydrocarbon scans of water and sediments at several places have also been run by the Corvallis laboratories of the EPA and have not revealed critical levels at any site, including the outlet of the dredge pipe and the outfall of the decanting lagoon. Scans of whole body samples of 20 white suckers for chlorinated hydrocarbons have also revealed no actionable level for any of the potentially troublesome materials. These analyses have been performed by the New York State Department of Environment Conservation.

PHOSPHORUS BUDGET

The inputs and outputs of phosphorus from the lake suggest that more phosphorus now leaves the lake than enters it (Table 2, after D. Howie). This supports the premise that phosphate, especially in the summer, is being released from the sediment.

One value that may be incorrect is that for dry and wet fall phosphorus into the lake. A value of 59 mg/m^2-yr was used, derived from the New York State Department of Public Health. This appears very high since this would give a concentrations of 66 μg/l for rain water (assuming all the phosphorus came in the form of rain). A lower value for atmospheric input would seem reasonable, thus decreasing the total input of phosphorus into the lake.

In the future, a mathematical model derived by Phillip D. Snow will be applied to the lake. Essentially the model is based on a mass balance incorporating input of phosphorus, sedimentation reactions and a rate of release of phosphorus from the sediment governed by the concentration of phosphorus in the interstitial water. All parts of the system (input, sedimentation, release) determine the ultimate concentration of phosphorus in the lake water. From preliminary studies, the model appears to be consistent with the results of Howie's paper in that release of phosphorus from the sediment, especially during the summer, appears to account for a significant amount of phosphorus in the water column. This is evident when one looks at summer ground water inflow (at 8 μg/l-P) and lake outflow (at 27 μg/l-P)(Table 2; D. Howie).

PHYTOPLANKTON

Pediastrum simplex is the predominant phytoplankter for much of the year. In the summer months of 1976 it was present at densities more than 1 × 10^6 cells per liter. The number of cells and not the number of clones is counted. The diatoms *Asterionella formosa*, *Melosira granulata* and *Synedra* sp. are also common but emerge more strongly during the fall and spring. The chlorophytes *Oocystis pusilla*, *Crucigenia quadrata*, *Scenedesmus quadrata*, and *Cryptomonas pusillus* are well represented during summer months. *Dinobryon bavaricum* and *Ceratium hirundella* are especially evident in the summer months as well. An increase in several species has been noted during and following dredging.

MACROPHYTES

The predominant aquatic macrophyte of the lake is the curly leaved pondweed, *Potamogeton crispus*. This species, introduced to America from Eurasia, grows to depths of three meters and occupied about 50% of the lake bottom prior to the beginning of dredging. At the peak of the growing season the lushly populated sectors demonstrated stem densities between 400 and 500/m^2 with oven dry weights of more than 100 g/m^2. The leafy stems rise from feeble rhizomes and reach to the surface to greatly stabilize water movement and to absorb plant nutrients. Under such conditions, Secchi disc readings reach their annual maximum of about 6 m.

In early June, turions are produced by the lateral branches and by the end of June, with the senescence of the parent plant, are released to float away, settle and reestablish new plants in August. These prosper during the fall, continue to grow under the ice and then with the melting of the ice cover grow vigorously to again reach the surface. The senescent plants are quickly consumed by various herbivorous invertebrates and bacteria, releasing their contained phosphorus and other plant nutrients to the water column. As a result, phytoplankton, notably *Pediastrum simplex*, becomes resplendent, reducing Secchi disc readings to a meter or less. The roots of the pondweed also degenerate at senescence and no nutrient translocation of the substrate typical of more perennial forms such as *Nymphaea* and *Peltandra* occurs.

As has been mentioned, the water chestnut, *Trapa natans*, has been prominent in the past but today only a few bushels of nuts may be collected from the entire lake.

Some 14 other species of macrophyte are also evident. Dredging has eliminated the pondweed from about 30 percent of its previous extent in the lake and under-ice observations by divers also indicate the absence of turions in one dredged sector where they were present the previous year.

ZOOPLANKTON

The rotifers are well represented by strongly seasonal appearances of *Keratella cochlearis*, *Kellicotia longispina*, *K. bostoniensis*, *Polyarthra* sp., *Filinia longiseta* and *P. euryptera* and the maxima of their populations occur in April through August.

The calanoid copepod *Diaptomus birgei* is most abundant in the spring, reaching densities of 50 to 100 per liter, but is present year around. Earlier, we had difficulty in identifying this uncommon form but now feel confident on the basis of well displayed fifth legs of males. *Mesocyclops edax* is the prevailing adult cyclopoid copepod of the summer while adult *Cyclops bicuspidatus thomasi* is the common form during the winter and spring months. The raptorial and predatory cyclopoid copepods may play an important role in regulating other smaller zooplankters. The large cladoceran *Daphnia galeata mendotae* is abundant at upper levels in the warmer months and in the depths during November and December. *Daphnia parvula* has appeared newly on August 3, 1977. After the beginning of dredging and, subsequently, it has become abundant, possibly replacing *D. galeata mendotae*. *Eubosmina coregoni*, one of the smaller cladocera, is the most abundant summer zooplankter, reaching peak densities of 250/l in July. *E. longispina* has proliferated during the fall of 1977, reaching densities greater than previously noted, and *Ceriodaphnia reticulata* is abundant in June, especially in the littoral zone.

The species listed are those prominent at station W, the most limnetic of the four study areas. Plankton samples made in the proximity of macrophytes and other substrates may be expected to contain many other species such as *Chydorus sphaericus*, *Monostyla closterocerca*, *Lepadella ovalis* and *Cephalodella* sp.

FISHES

The lake has a vigorous and diverse fish fauna. Bluegills, pumpkinseeds, black crappie and largemouth bass are the conspicuous centrarchids. Carp and suckers probably constitute the majority of biomass. Chain pickerel and a few nothern pike attract sport fishermen. The gizzard shad and alewife are occasionally netted and reflect the connection of the lake and the river.

The yellow perch and golden shiner are the most frequently and easily taken with gill net and have thus been elected for routine monitoring of qualities such as condition, gonadosomal and hepatomosomal indices, which are believed to reflect the health of their populations. The abundance and size of macrophage centers of liver, spleen and kidney are also assessed toward sensing physiological condition. At present, the yellow perch demonstrate condition indices of normal range and conventional gonadal development; however, reproduction is not evident, perhaps because of the inadequacy of existing spawning sites. The golden shiner population also appears vigorous and healthy, and young of the species are evident. Macrophage centers are large and abundant in the kidney and spleen of both the yellow perch and the golden shiner, and in the liver of only the yellow perch. But at this time we are unable to evaluate the significance of these observations.

PROGRAM PARTICIPANTS

The three senior investigators have had the profound satisfaction of working with a large number of exceedingly talented and dedicated student participants. Most of these workers have been juniors or seniors and they have commonly shouldered responsibilities and have demonstrated initiative appropriate for graduate work. Most of the data reported here is the product of their labors. We are very proud of them.

CIVIL ENGINEERING MAJORS

Physical-Chemical Analyses:

Daniel Berg 76-77; Andrew Adriance 76; Garvin Wells 76; Michael Sullivan 76; Timothy Pangburn 76-77; Mark Mossey 76-77; Steven Gyory 76; June Rinkoff 77-78; David Cornell 77-78; Timothy Baldwin 78; Philip Hood 78

Computer Analysis:

Rodney Aldrich 77; Douglas Howie 77-78; David Jones 77-78

BIOLOGY MAJORS

Phytoplanktonology;

Marian Baciewicz 76-77; Gloria O. Zorka 77-78; Joyce Warner 77-78;

Zooplanktonology:

Katherine Hollister 75-76; Adam Berg 76; Robert Lein 77-78; Judith Haddad 77-78; Bruce Levine 77-78;

Macrophyte Studies:

Paul Powers 76; Edward Heyes 76; Gary Geller 76; Joan Gebhardt 77-78; Alan Woodard 77-78;

Ichthyology-Histology:

Thomas Engel 75; Larry Merlis 75; Arnold Brender 76; Denise Polsinelli 77; Glenn Delaney 76-77; Carol Grant 76-77; Steve Lacy 76-78; Chris Brown 77-78; Carmen Gatta 77-78; Marc Klemperer 77-78; Terri Moran 77-78; Iain Drummond 77-78; John Foehl 77-78;

Computer Analysis:

James Lerner 77-78; David Powers 77-78

We also wish to thank several consultants whose good guidance has been especially helpful, namely Dr. Paul Mason for computer analysis, Dr. Wolfgang Fuhs and Dr. Helen Birecka for chemical analysis, Dr. Ed Mills and Ms. Susan Allen for plankton analysis, Dr. Eugene Ogden for macrophyte studies, Dr. Carl Schofield for ichthyological matters, Drs. Willard Roth, Abraham Rajender and George Smith for histology, Dr. Jay Bloomfield and Mr. Frank Stay for administrative *savoir-faire* and, finally, but most enthusiastically, Mr. Calvin Welch, Chairman of the Scotia Board of Park Commissioners, the person who "*got and kept things going*"!

TABLE 1. THE ANNUAL CYCLE OF TEMPERATURE, OXYGEN CONCENTRATION AND PERCENT SATURATION FOR STATION W AT 1 M DEPTH, COLLINS LAKE, DEMONSTRATING PRINT-OUT FORMAT.

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?101277

TYPE THE FIRST LETTER OF THE STATION YOU WANT PLOTTED SUCH AS W, E, OR G.
?W

TYPE THE DEPTH OF THE DATA YOU WANT TO HAVE PLOTTED.
?1.0

YOU WANT STATION W AND DEPTH 1.0 FROM SEP 14, 1976 TO OCT 12, 1977.  IS THIS
RIGHT.  ANSWER YES OR NO.
?YES

   DATE                  TEMP         DISS OXYGEN         PER CENT SAT
SEP 14, 1976             18.7            12.0                127.9
OCT 10, 1976             12.7             8.8                 82.7
OCT 28, 1976              5.7            12.4                 98.5
NOV 10, 1976              3.5            14.5                108.7
JAN  5, 1977              3.0            13.5                 99.8
JAN 19, 1977              3.0            13.8                102.0
FEB  2, 1977              2.0            10.0                 71.9
FEB 16, 1977              1.6             7.9                 56.2
MAR  2, 1977              3.0            12.3                 90.9
MAR 17, 1977              4.0            15.0                114.0
MAR 30, 1977              7.0            14.5                119.1
APR 13, 1977             11.4            15.3                139.6
APR 27, 1977             11.8            13.7                126.1
MAY 11, 1977             12.0            13.0                120.2
MAY 25, 1977             19.0            13.0                139.4
JUN  8, 1977             15.8             9.5                 95.4
JUN 22, 1977             20.4             7.9                 87.1
JUL  6, 1977             22.5             8.8                100.9
JUL 20, 1977             26.5            11.7                144.3
AUG  3, 1977             24.0            13.4                158.0
AUG 18, 1977             20.8            10.1                112.2
AUG 31, 1977             23.0            12.8                148.2
SEP 15, 1977             17.0             8.9                 91.7
SEP 28, 1977             14.9             8.0                 78.9
OCT 12, 1977             11.9             8.4                 77.5

DO YOU WANT TO PLOT THIS ON THE TYPEWRITER.  ANSWER YES OR NO.
INVALID RESPONSE.  TRY AGAIN.
INVALID RESPONSE.  TRY AGAIN.
?YES

DO YOU WANT TO PLOT TEMPERATURE (TE), DISSOLVED OXYGEN (DO), OR PER CENT SATU-
RATION (PC).  ?DO
```

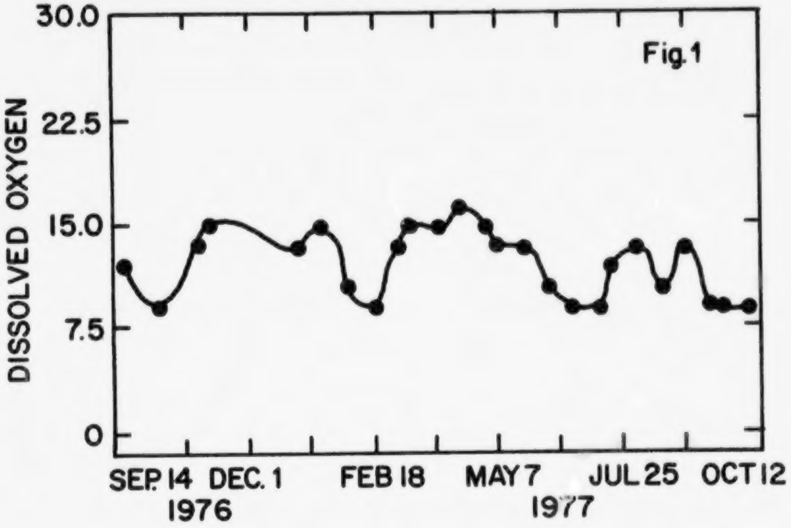
TABLE 2. A PRELIMINARY PHOSPHORUS BUDGET FOR COLLINS LAKE, NEW YORK (AFTER D. HOWIE).

Groundwater Input	12.8 Kg/yr
Mohawk River Input	20.4 Kg/yr
Dry and wet fall Input	14.3 Kg/yr
Dumped Snow Input	0.09 Kg/yr
Storm Sewer Input	0.04 Kg/yr
Total Input	47.63 Kg/yr
Outflow to Mohawk River	52.71 Kg/yr
New Difference Output-Input	5.48 Kg/yr

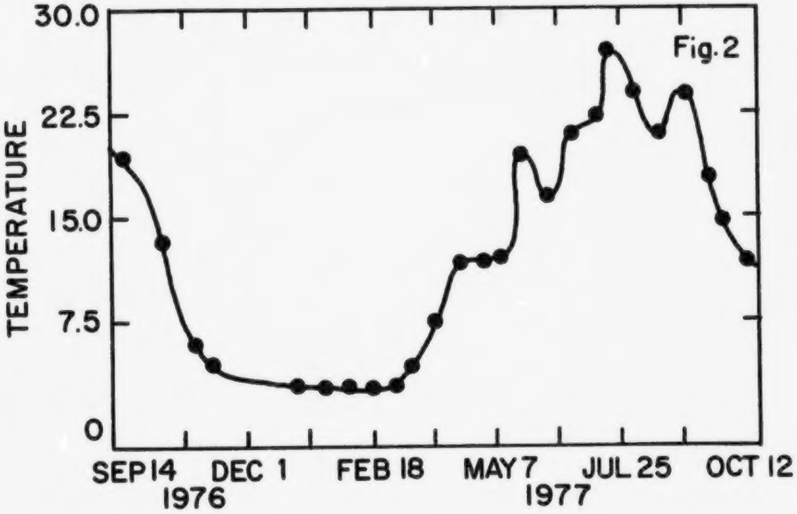
TABLE 3. PHOSPHORUS CONCENTRATIONS OF GROUNDWATER ENTERING THE LAKE AT SPRINGS ALONG THE NORTHERN SHORE AS COMPARED WITH THOSE AT THE LAKE OUTLET, COLLINS LAKE, 1977 (AFTER D. HOWIE).

	DATE	GROUNDWATER (μg/1P)	OUTLET (μg/1P)
1977	1-5	---	12
	1-19	---	8
	2-2	---	16
	2-16	---	10
	3-2	---	10
	3-17	---	29
	3-30	---	45
	4-13	---	8
	4-27	---	23
	5-11	---	12
	5-25	---	15
	6-8	9	25
	6-22	8	25
	7-6	8	32
	7-20	1	22
	8-3	1	26
	8-18	4	25
	8-31	8	34
	9-15	8	17
Yearly average		5	21
June-August average		8	27

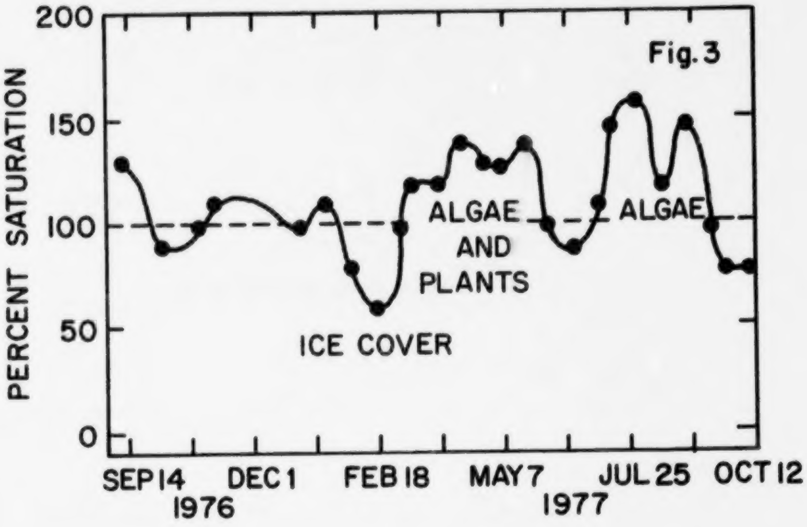
DISSOLVED OXYGEN W STA AT DEPTH 1.0



TEMPERATURE W STA AT DEPTH 1.0



PERCENT SATURATION W STA AT DEPTH 1.0



TEMPERATURE WEST STA AUG 3, 1977

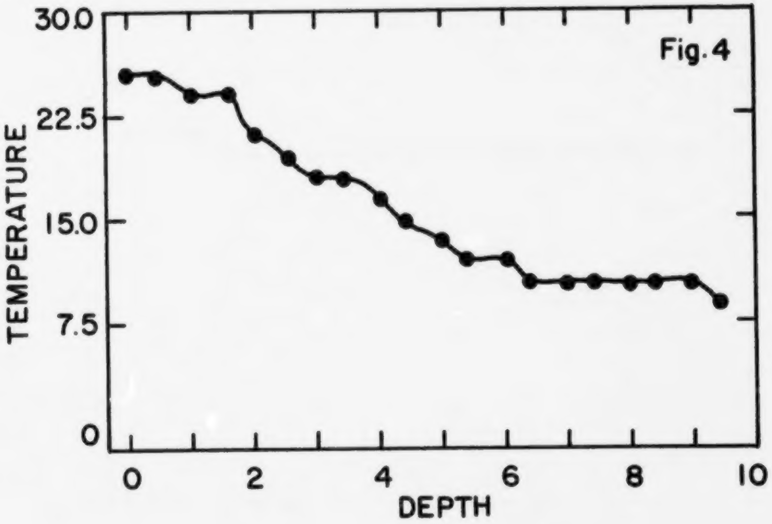


Fig. 5

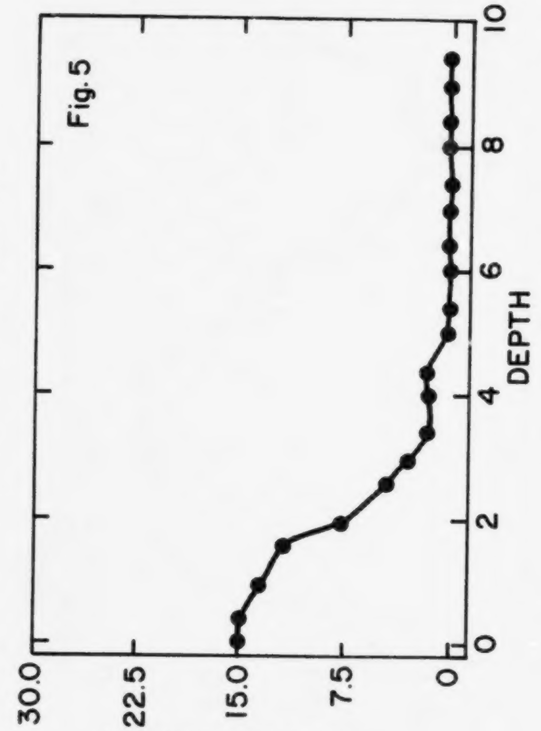

DISSOLVED OXYGEN STA AUG 3,1977

Fig. 6

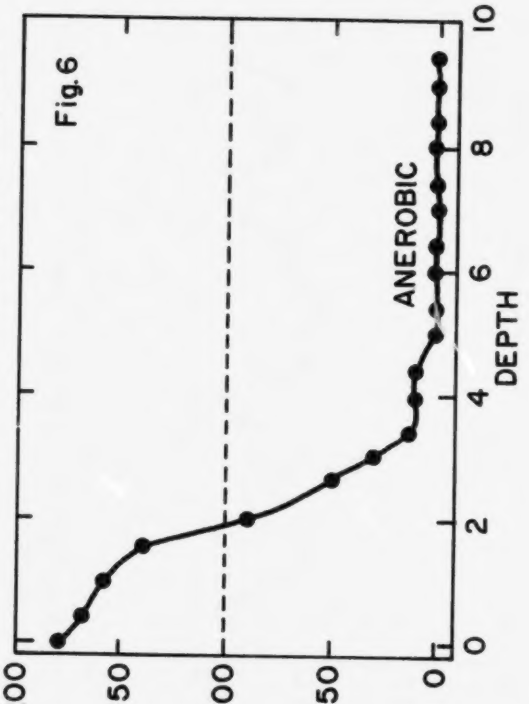

PERCENT SATURATE WEST STA AUG 3, 1977

Fig. 7

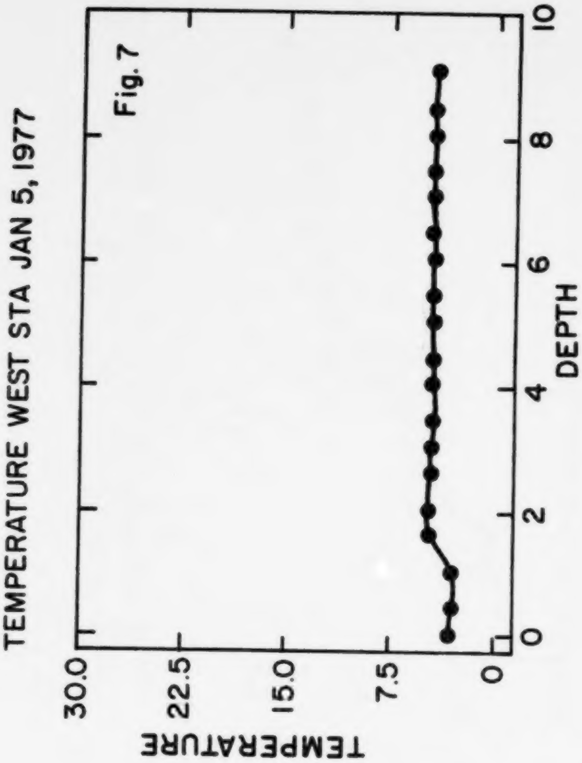
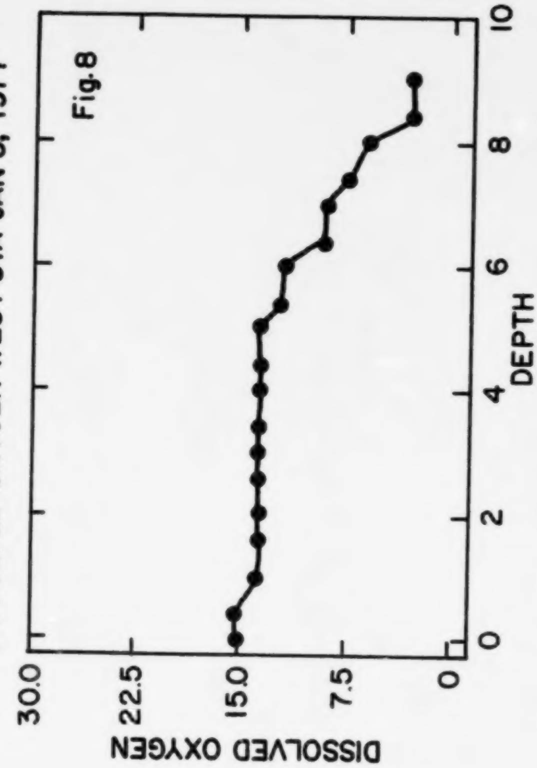
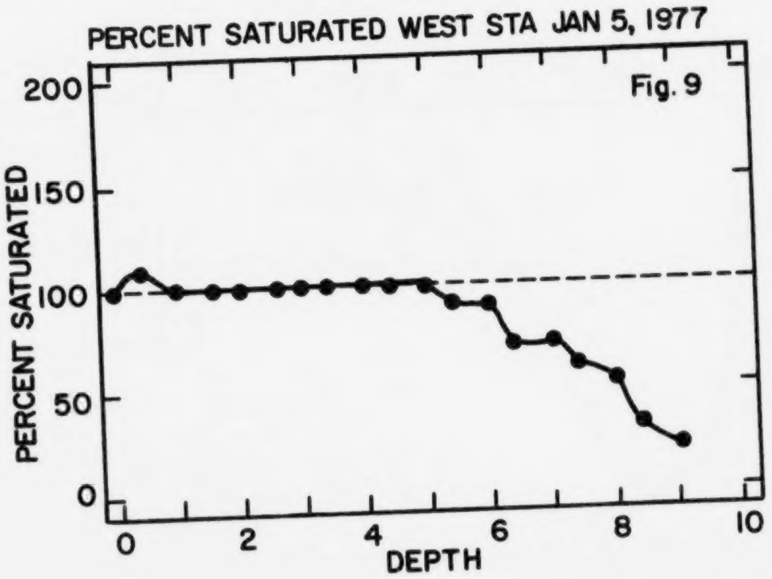
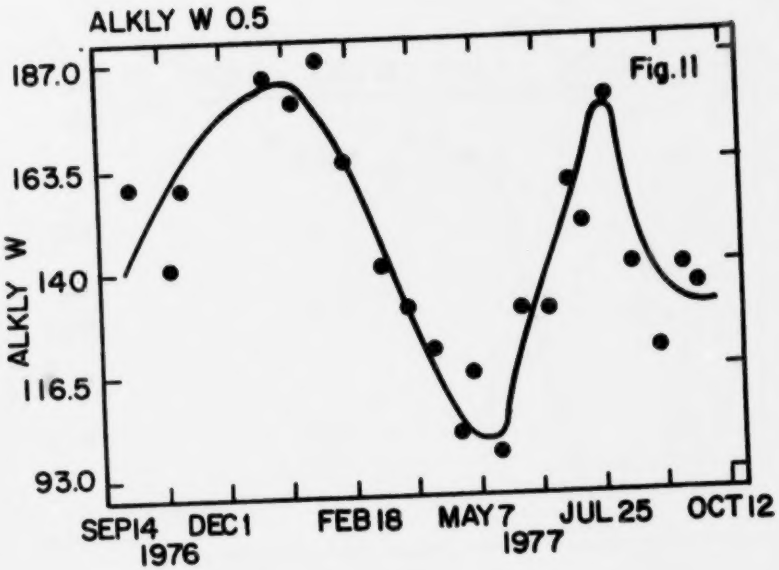
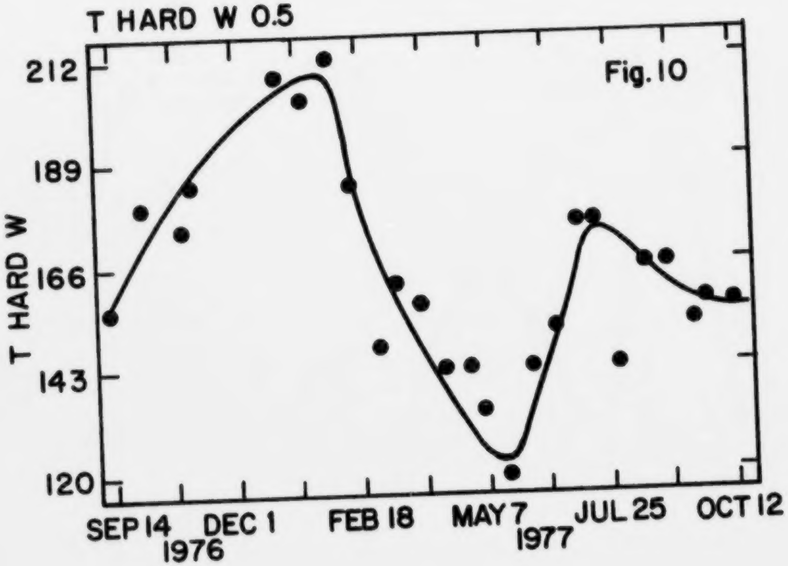

TEMPERATURE WEST STA JAN 5, 1977

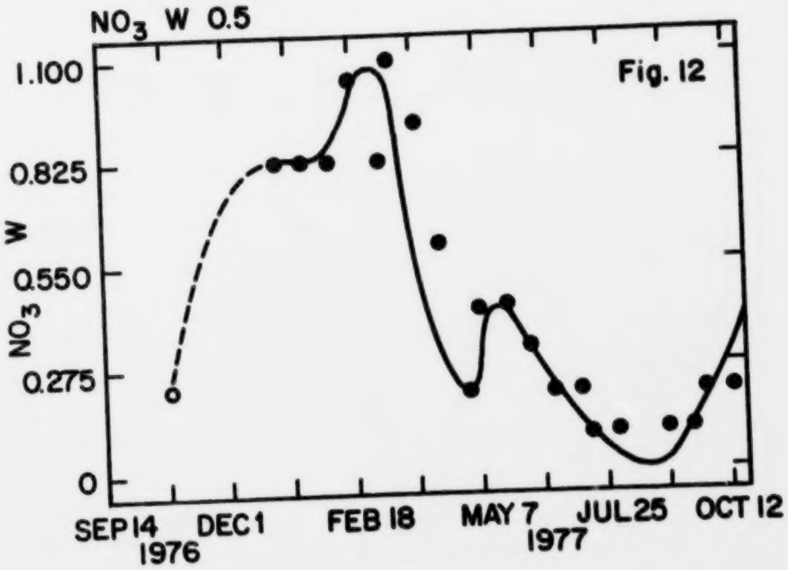
Fig. 8


DISSOLVED OXYGEN WEST STA JAN 5, 1977


PERCENT SATURATED WEST STA JAN 5, 1977
Fig. 9
PERCENT SATURATED
200
150
100
50
0
DEPTH
0
2
4
6
8
10


ALKLY W 0.5
Fig. 11
ALKLY W
187.0
163.5
140
116.5
93.0
SEP14 1976
DEC1
FEB18
MAY7 1977
JUL25
OCT12


T HARD W 0.5
Fig. 10
T HARD W
212
189
166
143
120
SEP14 1976
DEC 1
FEB 18
MAY7 1977
JUL 25
OCT 12


NO3 W 0.5
Fig. 12
NO3 W
1.100
0.825
0.550
0.275
0
SEP14 1976
DEC1
FEB 18
MAY 7 1977
JUL25
OCT 12

NH_3 W 0.5

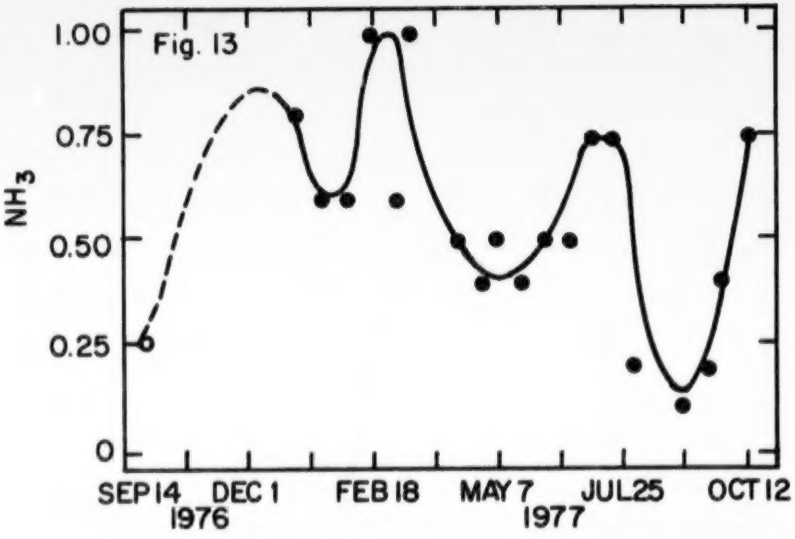


Fig. 13

NH_3 W 8.0

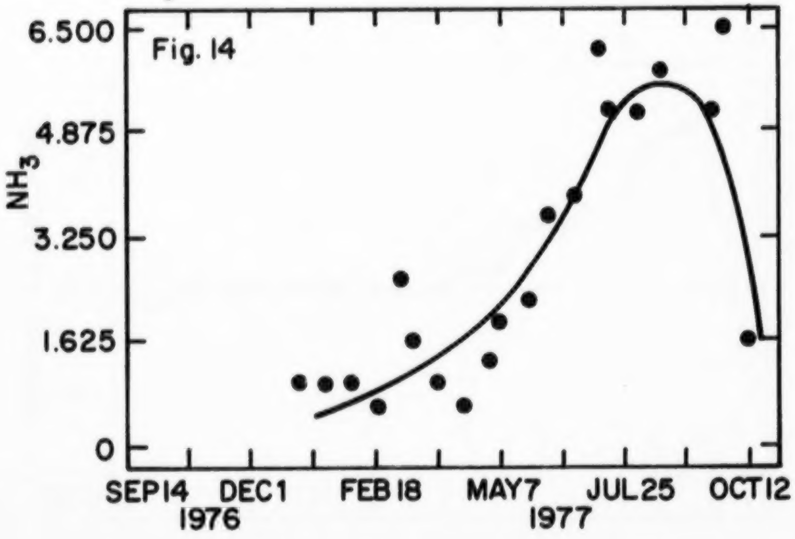


Fig. 14

T PHOS W 0.5

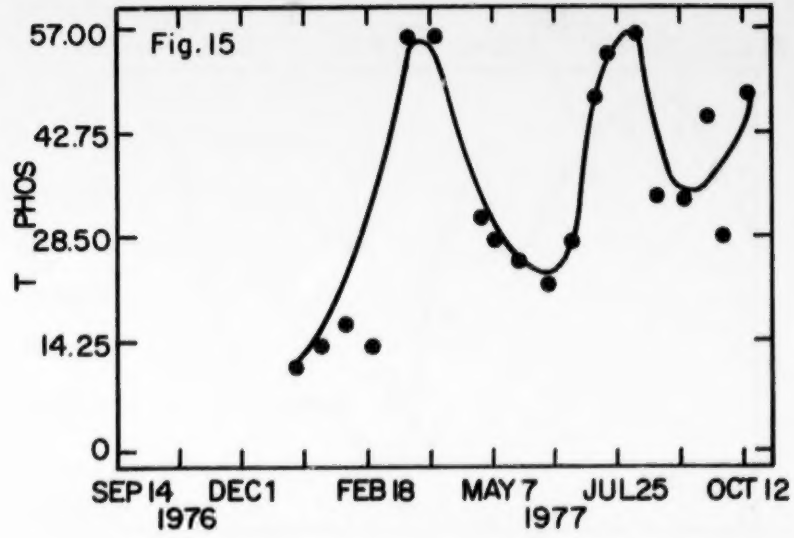


Fig. 15

T PHOS W 8.0

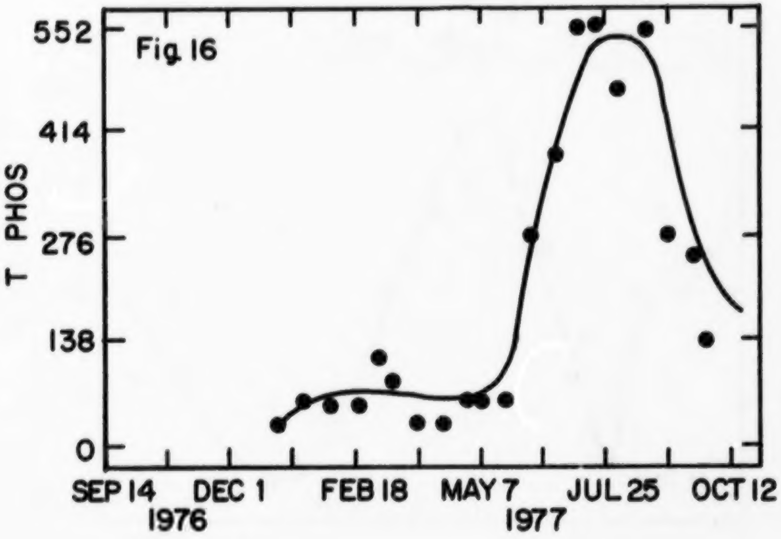


Fig. 16

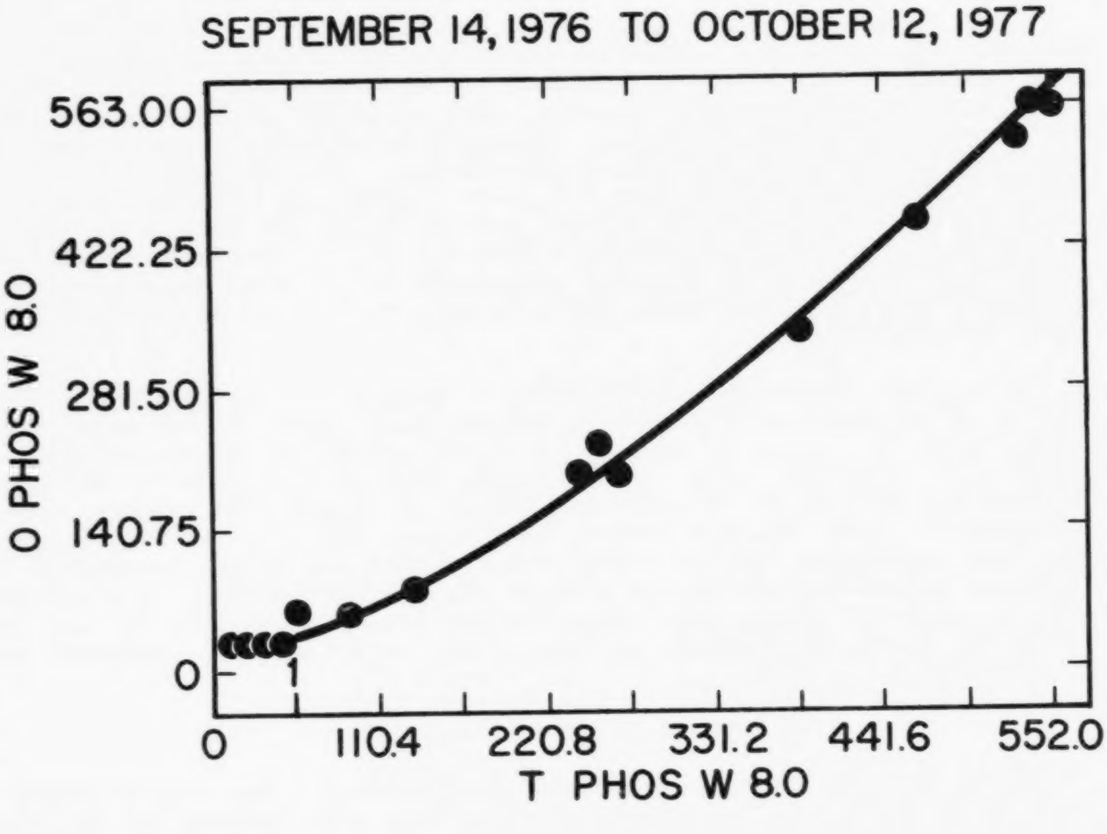

SEPTEMBER 14, 1976 TO OCTOBER 12, 1977
563.00
422.25
281.50
140.75
0
O PHOS W 8.0
1
0
110.4
220.8
331.2
441.6
552.0
T PHOS W 8.0

Figure 17.

EFFECT OF DREDGING AND NUTRIENT INACTIVATION AT LILLY LAKE, WISCONSIN

by

R. Dunst and R. Beauheim*

INTRODUCTION

Lake aging is a naturally occurring, continual process which eventually leads to lake extinction. Infilling can be caused by autochthonous, as well as allochthonous materials. In a lake with a small watershed, sediment influx may be minimal, but nutrients can enter via groundwater and/or surface runoff overly highly developed shorelines. The nutrient loading can support massive algal populations and/or dense macrophyte growths. The plant residual subsequently settles to the lake bottom and causes a reduction in depth. Decreased water depth is a problem in itself, and the shallower lakes allow rooted vegetation to invade a greater share of the lake basin. Lake use problems are intensified throughout the process, causing severe alterations and restrictions in recreation. Ultimately the basin will fill in and be converted into a dry land environment.

Lilly Lake is presently in an advanced stage of the aging process. The lake is 37 hectares in size with a maximum water depth of 1.8 meters, and greater than 10.7 meters of underlying organic sediments. Rooted macrophytes extend over the entire basin. Rehabilitation will involve hydraulic dredging, followed by application of aluminum sulfate if needed (EPA Project Number S804235-01-0). The physical effects of dredging are known and the various equipment, techniques and costs have been reviewed, but little information is available concerning biochemical effects on lake environments. Evaluations will be conducted during this rehabilitation project, and will include the effect on algae, macrophytes, benthos, fish, water quality, sediments and groundwater.

DRAINAGE BASIN CHARACTERISTICS

Lilly Lake is located in southeastern Wisconsin. The lake occupies a kettle-like depression in a topographically high area between the drainage basins of Palmer Creek, to the north and west of the lake, and Bassett Creek, to the south and east of the lake. Both of the streams flow northeasterly into the Fox River (see Figure 1). The lake is normally at an elevation of 230.6 meters (756.4 feet) above mean sea level. Hills to the north and south-

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* Office of Inland Lake Renewal, Wisconsin Department of Natural Resources, Madison, Wisconsin 53707.

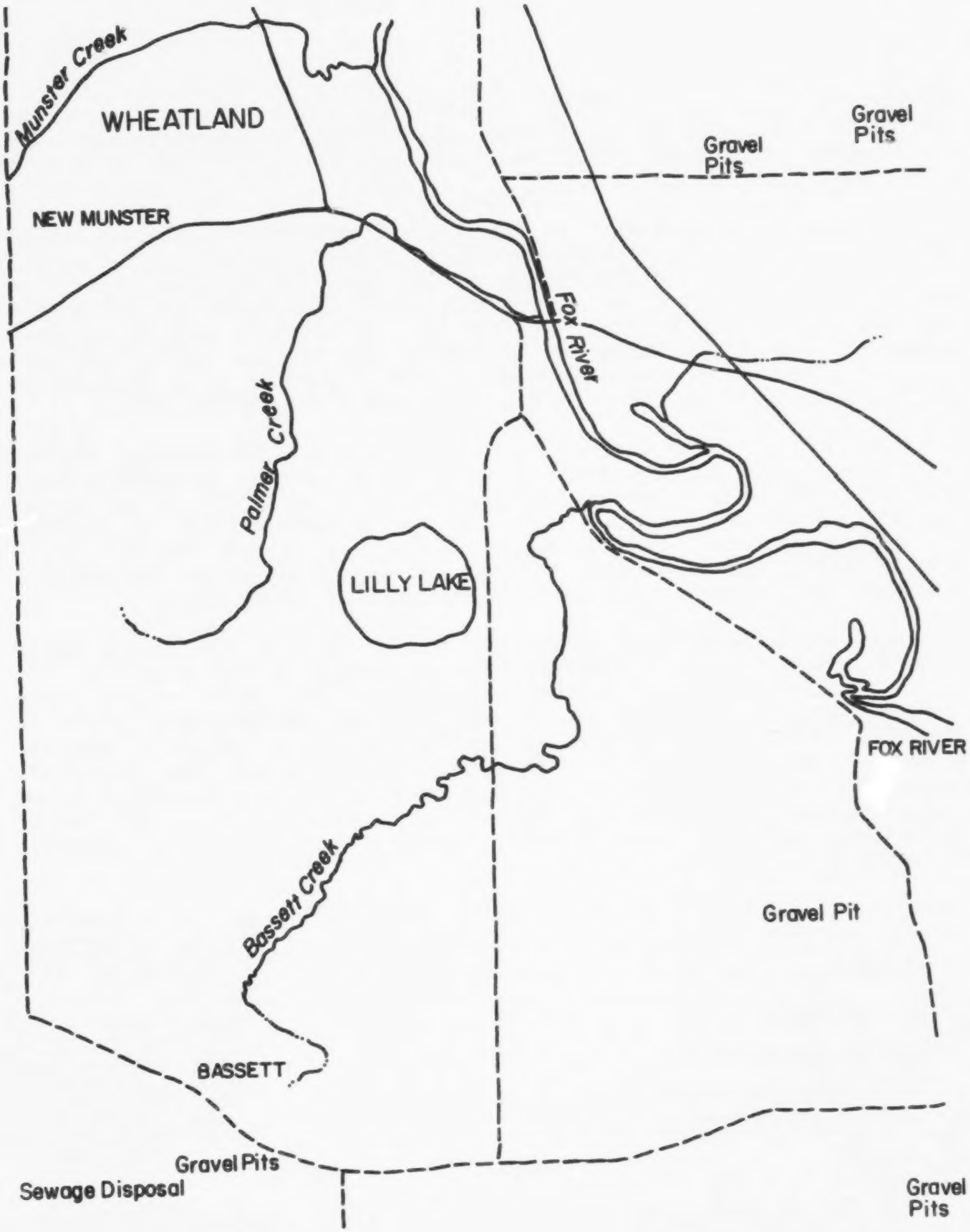

Munster Creek
WHEATLAND
Gravel Pits
Gravel Pits
NEW MUNSTER
Fox River
Palmer Creek
LILLY LAKE
FOX RIVER
Bassett Creek
Gravel Pit
BASSETT
Gravel Pits
Sewage Disposal
Gravel Pits

Figure 1.

west of the lake rise from 21.3 to 30.5 meters (70 to 100 feet) above the lake. The floodplains of Palmer and Bassett Creeks are at approximately an elevation of 228.6 meters (750 feet) and mean sea level. The terrain is gently rolling and irregular.

The area is underlain at depth by the Niagara dolomite and was subsequently covered by the Lake Michigan lobe of the Wisconsin glacial stage, which is primarly responsible for the surface deposits. The lake lies in an area of ice contact drift near one of the terminal moraines of the Lake Michigan lobe. The drainage pattern in the vicinity of Lilly Lake is somewhat irregular, showing influences of glaciation and of more recent erosional activity. The soils in the area are Miami loam, Miami fine sandy loam, Rodman gravelly loam, peat, Fox silt loam, Miami silt loam and Clyde silt loam.

The watershed which contributes to Lilly Lake is 155.4 hectares, producing a watershed to lake size ratio of 4. This area receives during an average year 1.3×10^6 cubic meters of precipitation, of which 0.3×10^6 are direct rainfall on the lake surface. Lilly Lake has no surface inlets or outlets. The regional movement of groundwater is northeast toward the Fox River, roughly paralleling the surface drainage. At normal levels the lake contains 0.5×10^6 cubic meters of water and has a mean depth of 1.4 meters.

Sand predominates along 65 percent of the shoreline, gravel and rubble cover 6 percent, and soft material cover 29 percent of the shoreline. The entire center is composed of organic sediment with a high water content (Table 1). The dredging operation is designed to remove 596,353 cubic meters of sediment, increasing the maximum depth of the lake to 6.1 meters (Figure 2). Because of the high percent of water content of the sediments and their fluid nature, it will be possible to pump the sediments almost 3 kilometers to an inactive gravel pit. Plans also include some application of the material to agricultural lands via spray irrigation and/or low level flooding. Dredging is scheduled to begin later this spring.

TABLE 1. SOLIDS CONTENT OF THE SEDIMENTS: SEPTEMBER 20, 1977 (4 LOCATIONS)

Depth into Sediments	Percent Dry Solids	Percent Water
1.5 meters (5 ft)	2.4 - 3.6	96.4 - 97.6
3.7 meters (12 ft)	3.1 - 4.3	95.7 - 96.9
6.1 meters (20 ft)	4.8 - 9.4	90.6 - 95.2

EVALUATION

The project is intended to determine the overall effectiveness of dredging and nutrient inactivation at Lilly Lake. There will, however, be a special emphasis on the evaluation of existing predictive approaches. Data collection has and will be conducted before, during and after the rehabilitation program. The project period began March 28, 1977, although some information was already being gathered in 1976.

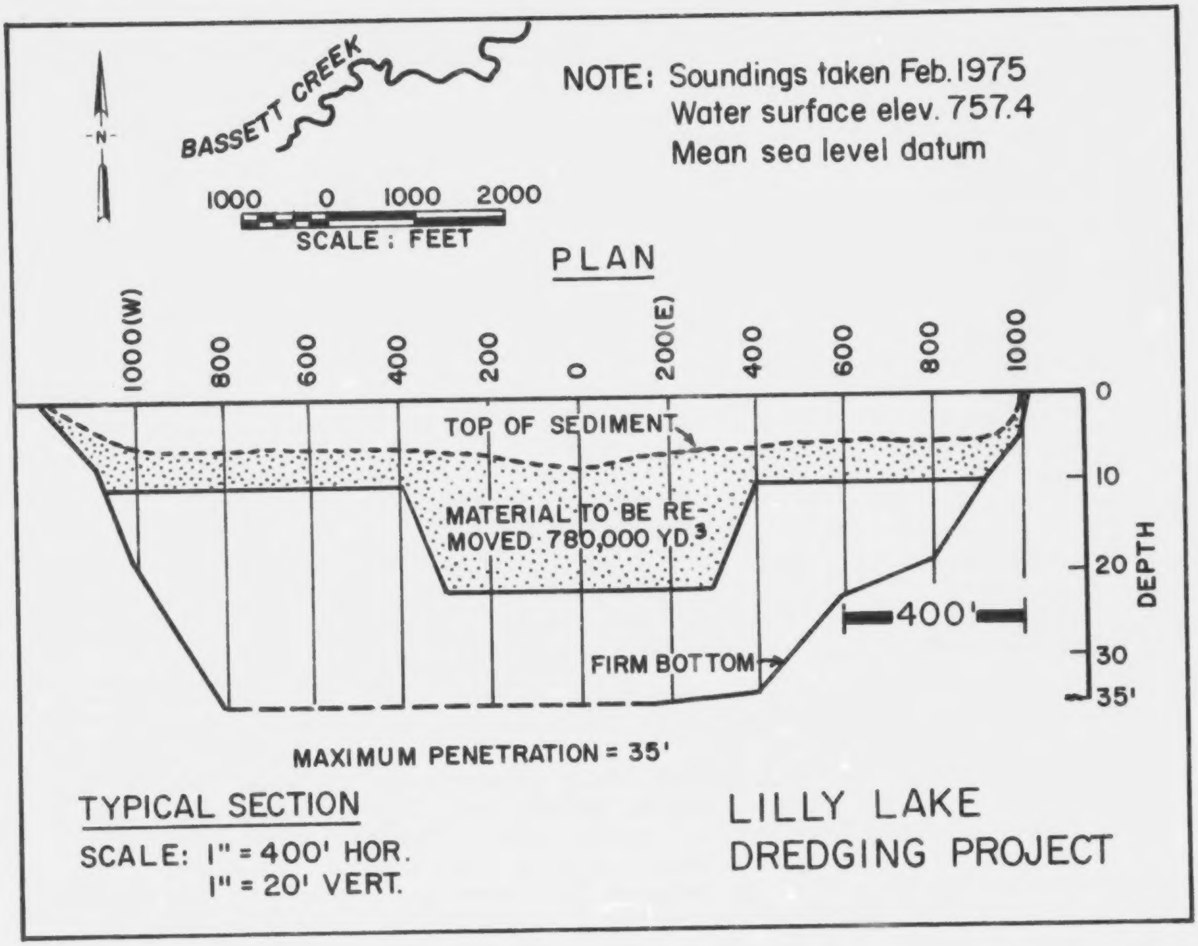

BASSETT CREEK
N
NOTE: Soundings taken Feb. 1975
Water surface elev. 757.4
Mean sea level datum
1000 0 1000 2000
SCALE : FEET
PLAN
1000(W)
800
600
400
200
0
200(E)
400
600
800
1000
TOP OF SEDIMENT
MATERIAL TO BE REMOVED 780,000 YD.³
400'
FIRM BOTTOM
0
10
20
30
35'
DEPTH
MAXIMUM PENETRATION = 35'
TYPICAL SECTION
SCALE: 1" = 400' HOR.
1" = 20' VERT.
LILLY LAKE
DREDGING PROJECT

Figure 2. Planned dredging project, Lilly Lake.

INLAKE PHYSICOCHEMISTRY

Temperature and dissolved oxygen profiles (one meter depth intervals) are being determined at a central location biweekly throughout the year, plus at four additional sites during the winter. Water samples are also being taken quarterly at these sites—spring overturn, summer stratification (August), fall overturn, and winter stagnation (February). The parameters are shown in Table 2. This information will permit evaluation of conditions in the new lake type.

TABLE 2. INLAKE WATER CHEMISTRY; SPRING OVERTURN

Parameter	4/28/75	4/6/76	3/15/77	4/11/77
Nitrite-N	.005	.004	---	.003
Nitrate-N	.08	.05	<.02	<.02
Ammonia-N	.09	.09	.02	<.04
Organic-N	.89	.69	.70	.61
Total-N	1.07	.82	.74	.67
SRP	.01	.01	<.004	.009
TP	.02	.03	<.02	.03
Calcium	34	28	23	25
Magnesium	28	13	11	14
Sodium	5	11	3	<1
Potassium	2.2	<.5	.2	<.5
Conductivity*	202	224	190	245
Sulfate	11	9	6	---
Chloride	5	5	5	8
pH (units)	8.0	8.0	7.8	7.8
Alkalinity	88	200	92	102
Turbidity	1.8	1.3	.5	4.1
Iron	---	<0.9	---	<.06
Manganese	---	<.03	---	<.03

* Expressed in micromhos/cm at 25°C; all other parameters are in mg/l.

One objective is to delineate, in a general way, the water quality effects of converting a shallow water, polymictic, littoral zone lake into a deep water, dimictic lake with a limnetic area. The depth of the thermocline will be compared with predictions based on past studies in nearby lakes. Oxygen depletion rates will be computed for summer and winter, and the results will be evaluated in terms of available oxygen depletion models (e.g. Veith and Conway, 1972). In addition, spring phosphorus concentrations will be compared with summer chlorophyll _a_ concentrations (Sakamoto, 1966; Dillon and Rigler, 1974). Thus far, results have shown the lake to be well mixed during the open water period with adequate dissolved oxygen concentrations. The shallow water allows rapid heating and cooling, and near noon temperatures of 30°C have been noted. During the winter of 1976-77, the dissolved oxygen levels became sufficiently low to result in a severe fishkill.

The sampling program also includes a measurement of BOD (5 day), turbidity, SRP, TP, nitrate/nitrite-N, ammonia-N and organic N on a biweekly basis. These are being measured to determine the immediate and subsequent short term effects of each treatment. The results for the May/September, 1977 period are shown in Table 3.

TABLE 3. INLAKE WATER CHEMISTRY; MAY-SEPT., 1977

Parameter	Range	Average
Nitrite/Nitrate-N		<0.02 mg/l
Ammonia-N	<0.02 - 0.08 mg/l	0.03 mg/l
Organic-N	1.0 - 1.7 mg/l	1.4 mg/l
SRP	<4 - 6 µg/l	4 µg/l
TP	20 - 70 µg/l	34 µg/l
Turbidity	0.6 - 1.9 mg/l	1.1 mg/l
BOD (5 day)	1.3 - 2.4 mg/l	1.8 mg/l

PHYTOPLANKTON

The evaluation studies include measurement of chlorophyll a, Secchi disc, and primary productivity. Sampling is being conducted biweekly May through October at one meter depth intervals from a central location. Productivity is being measured by the light and dark bottle dissolved oxygen method— noon to 6 p.m. The inlake DO levels were also noted at the start and end of the period.

The spring TP concentrations have ranged from 20 to 30 µg/l (Table 2), and the N:P ratio has always exceeded 20. According to Figure 3, average summer chlorophyll a concentrations of 5.6 to 10 µg/l would have been predicted. However, the June/August averages were 2.5 and 3.0 µg/l for 1976 and 1977, respectively (Figure 4). The highest value in either year was only 6 µg/l. During these same periods, primary productivity (phytoplankton) was 18 and 12 µg C/l/hr, respectively. Primary productivity and chlorophyll a levels tended to fluctuate together but the statistical relationship was poor. Water clarity measurements have always exceeded 1.8 meters, the maximum depth of the lake.

Between noon and 6 p.m. the inlake dissolved oxygen concentrations always increased by an amount far above that possible due to phytoplankton productivity alone. Because the lake is shallow and well mixed, the water temperature would normally be increasing during this period, oxygen transfer into the lake across the air-water interface is a doubtful causative factor. The inlake dissolved oxygen increase is more likely due to the photosynthetic activity of attached plants, primarily macrophytes. The increase is therefore presented as a community primary productivity (net) when compared with phytoplankton productivity (gross) in Figure 5. The phytoplankton may be inhibited by the extensive macrophyte growths. Dredging may, however, greatly reduce the littoral zone and the newly created limnetic area may support the algal growths predicted from Figure 3.

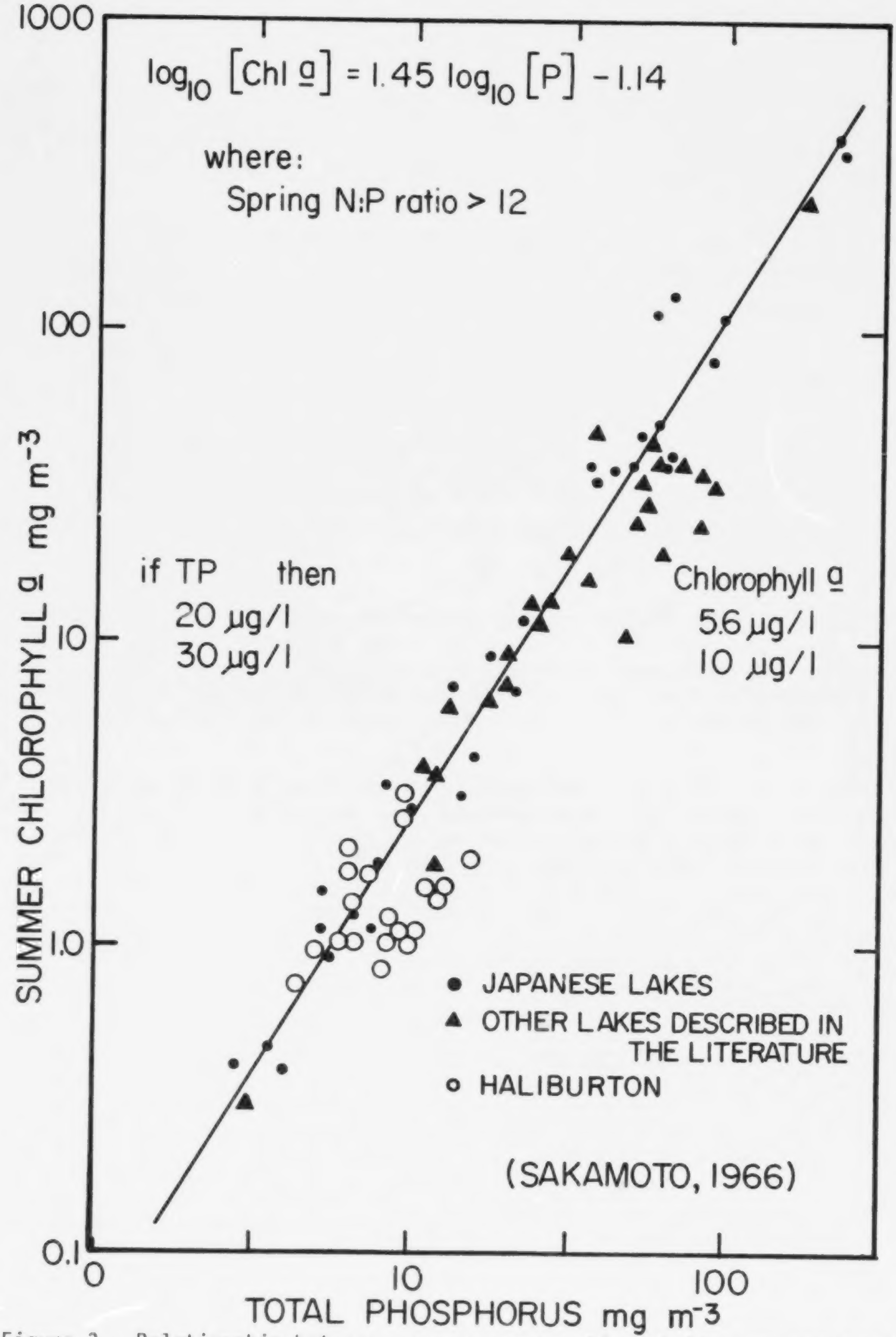


Figure 3. Relationship between average summer chlorophyll a and spring TP.

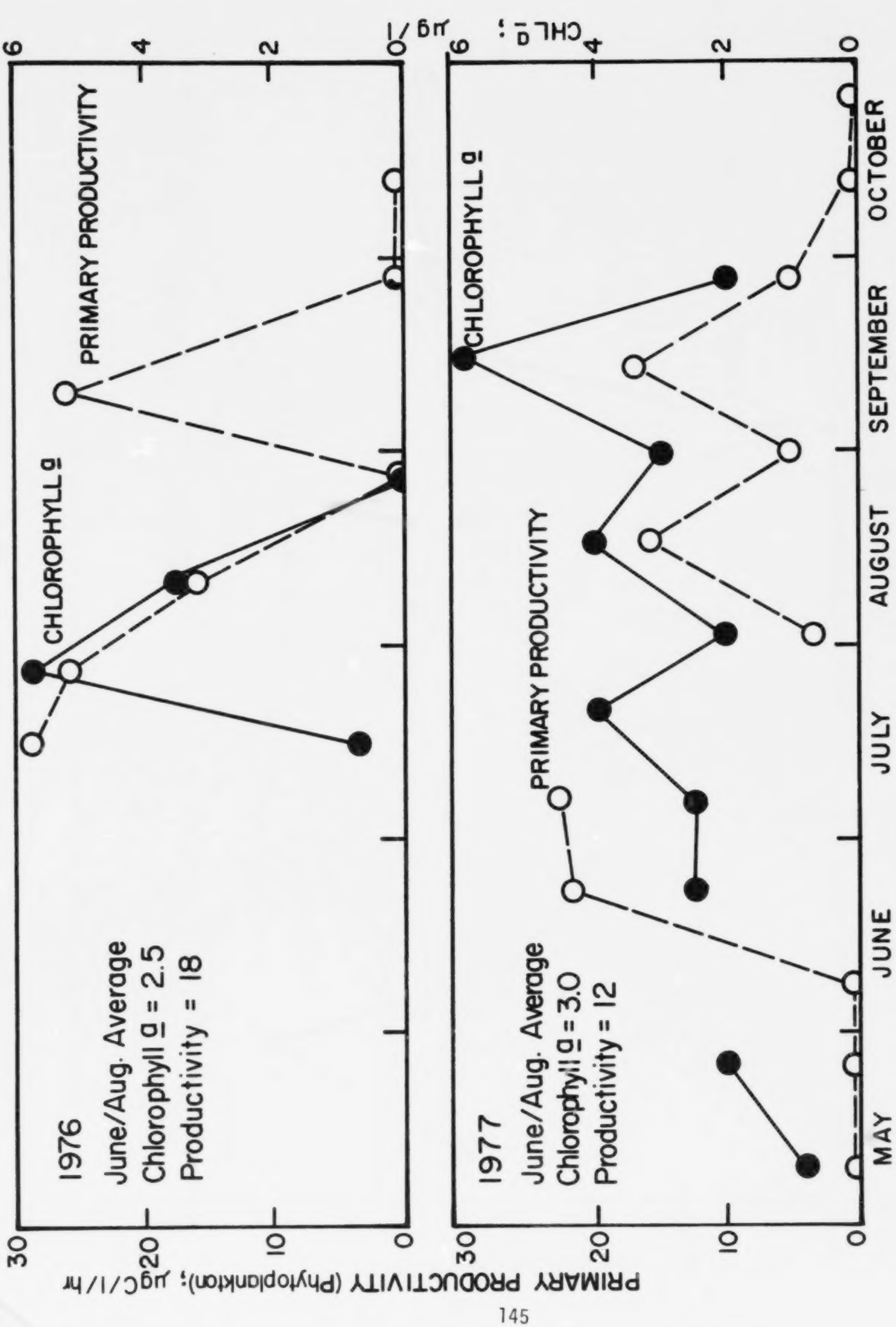


Figure 4. Phytoplankton productivity and chlorophyll a during 1976 and 1977.

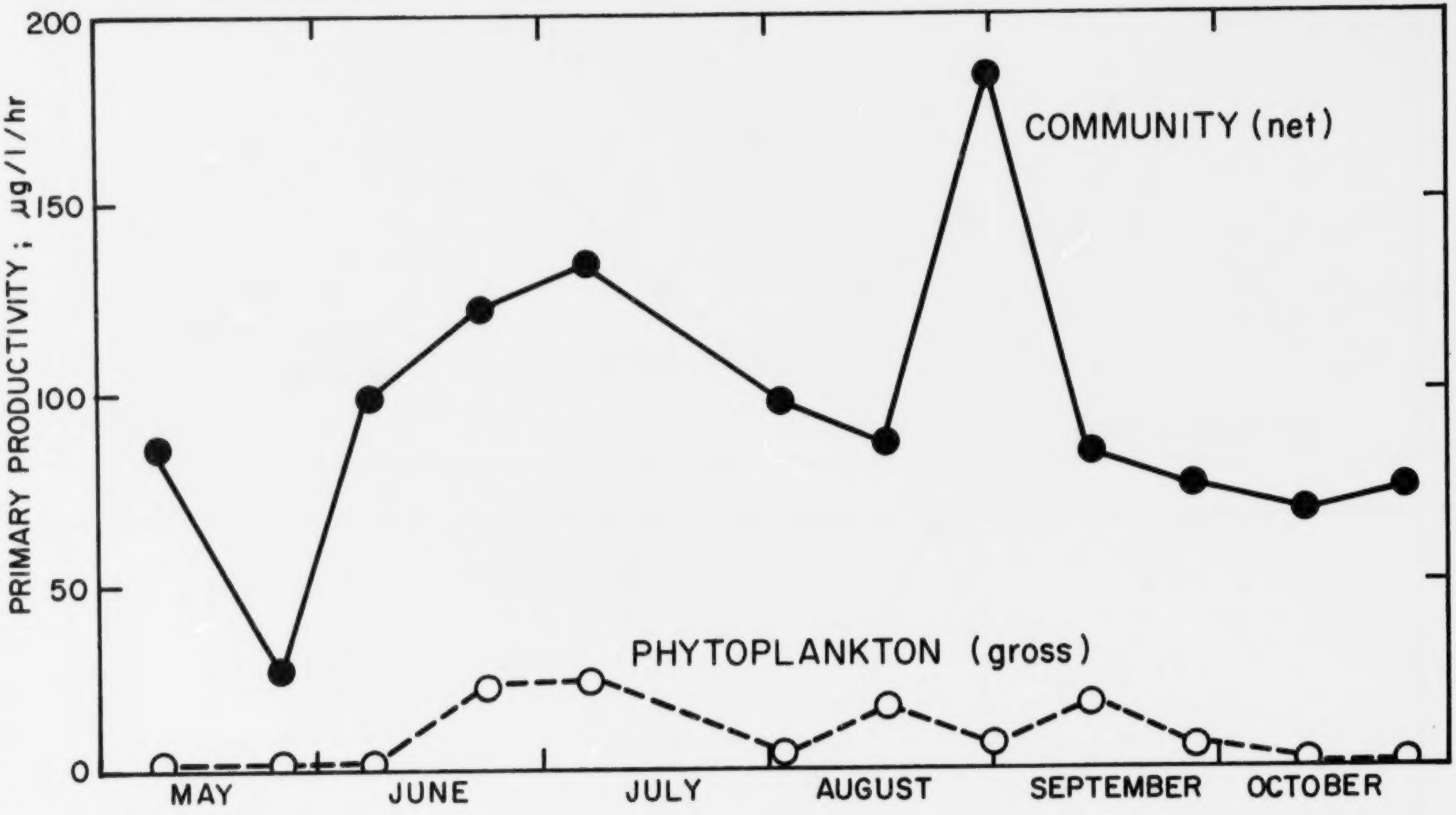


Figure 5. Comparison of primary productivity in 1977—community vs. phytoplankton.

MACROPHYTES

The macrophytes are being sampled biweekly, May through September. Ten 0.1 square meter biomass samples are being removed by diving from each of 10 areas per sampling date. These are placed in plastic bags, brought back to the laboratory, cooled overnight, and processed the next day. Drying is accomplished at 105°C.

The lake will be deepened to a maximum of 6.1 meters, with dredging down to "hard bottom" in the near shore areas. It is anticipated that predominantly a sand/silt bottom will be created to a depth of approximately 3 meters. The remainder of the lake basin, from 3 meters down to 6.1 meters, will contain organic sediments. The evaluations include: 1) comparison of maximum depth of growth versus predictions based on water clarity, 2) examination of the hard bottom (newly exposed) community versus predictions, and 3) investigation of the relationship between phytoplankton and macrophytes.

In terms of water clarity, if the present 2.5-3.0 μg/l of chlorophyll *a* persists after dredging, the maximum depth of growth should be 4.6-5.2 meters (Figures 6 and 7). Most of the mid-lake area would therefore contain macrophytes. However, if chlorophyll *a* values approach 10 μg/l after dredging, the whole area might be macrophyte free.

The weighted average biomass for the lake as a whole was 620 g/m^2 in early 1976; subsequently, it gradually increased to 840 g/m^2 by late August, and then dropped sharply in early September. In 1977 it was 540 g/m^2 in mid-May and slowly decreased through most of the summer to 250 g/m^2 by mid-September (Figure 8). *Potamogeton Robbinsii* is by far the most important species in terms of biomass, followed by *P. amplifolius* and *P. praelongus*. All three of these species, and especially *P. Robbinsii*, overwinter at relatively high densities. In general there appeared to be an inverse relationship between chlorophyll *a* and macrophyte biomass (Figure 9). This was true in both years, although for a given macrophyte biomass, lower chlorophyll *a* concentrations would have occurred in 1977. The average summer TP levels were 120 μg/l (1 sample) and 36 μg/l (6 samples; range of 20-70 μg/l) in 1976 and 1977, respectively. Although TP levels tended to increase during the summer as the macrophytes were declining, there was no precise relationship (Figure 10). Macrophyte samples have been retained for tissue analyses, but assuming a content of 0.13 percent P, 100 g/m^2 of macrophytes would be equivalent to an inlake P concentration of 96 μg/l at normal water levels.

In terms of conversion of soft bottom to hard bottom there are 3 depth zones of interest—0 to 0.5 meters, 0.5 to 1.5 meters and 1.5 to 3 meters. When the 0-0.5 meter zone is converted from soft to hard bottom, about eight species will be greatly reduced (Table 4). The only species expected to increase are *P. illinoensis*, *Chara* spp., and *Najas flexilis*. At the 0.5-1.5 meter depth, about nine species are expected to decrease, with *P. amplifolius*, *P. illinoensis*, *Chara* spp., and *Najas flexilis* increasing (Table 5). The reduction in biomass in the 0-1.5 meter zone undergoing conversion to hard bottom will be at least 70 percent (Table 6). Table 6 also points out that the lower biomass in 1977 versus 1976 was due to a 50 percent reduction in the soft bottom areas. Biomass actually increased over hard bottom. According to

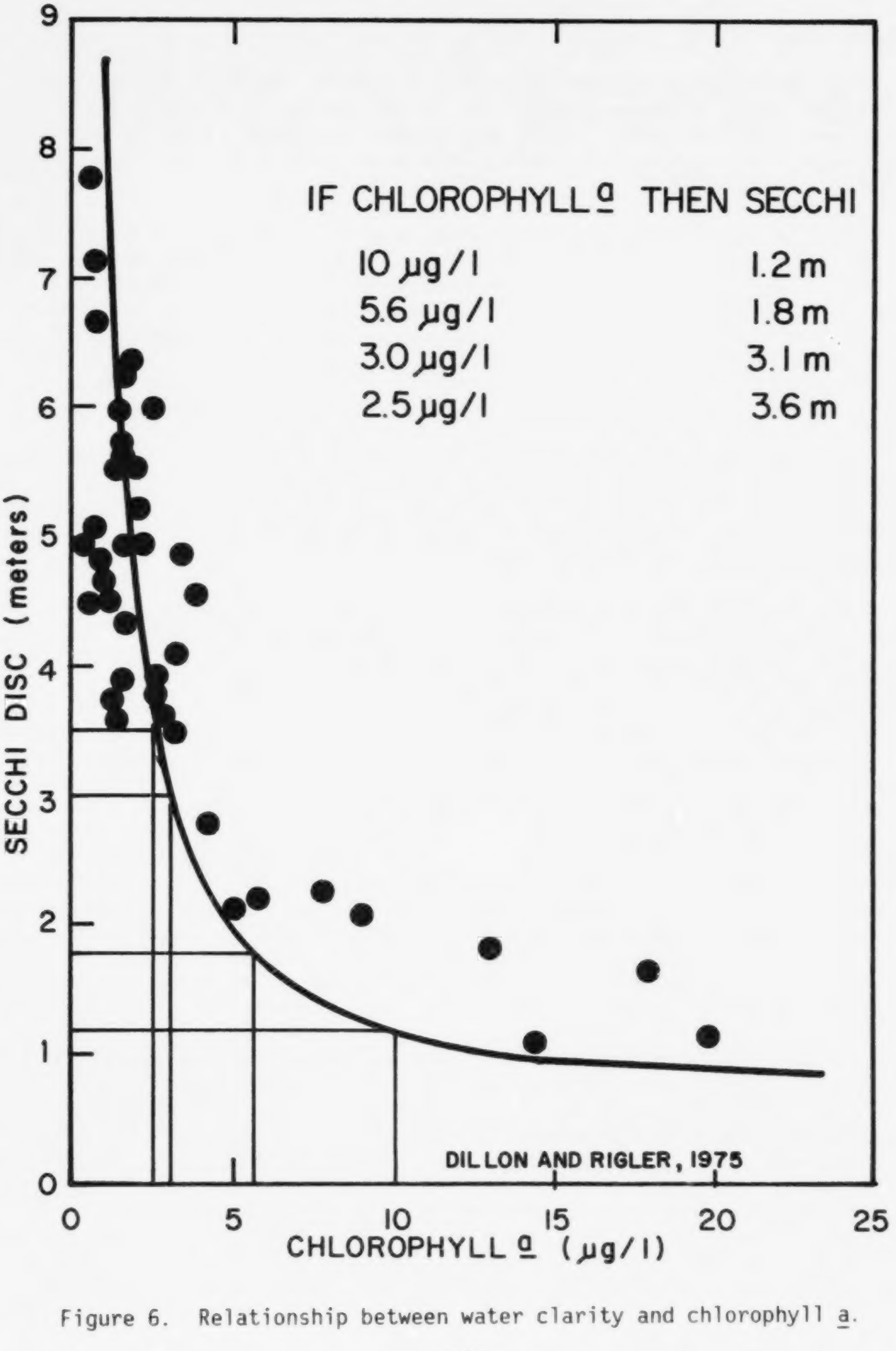


Figure 6. Relationship between water clarity and chlorophyll a.

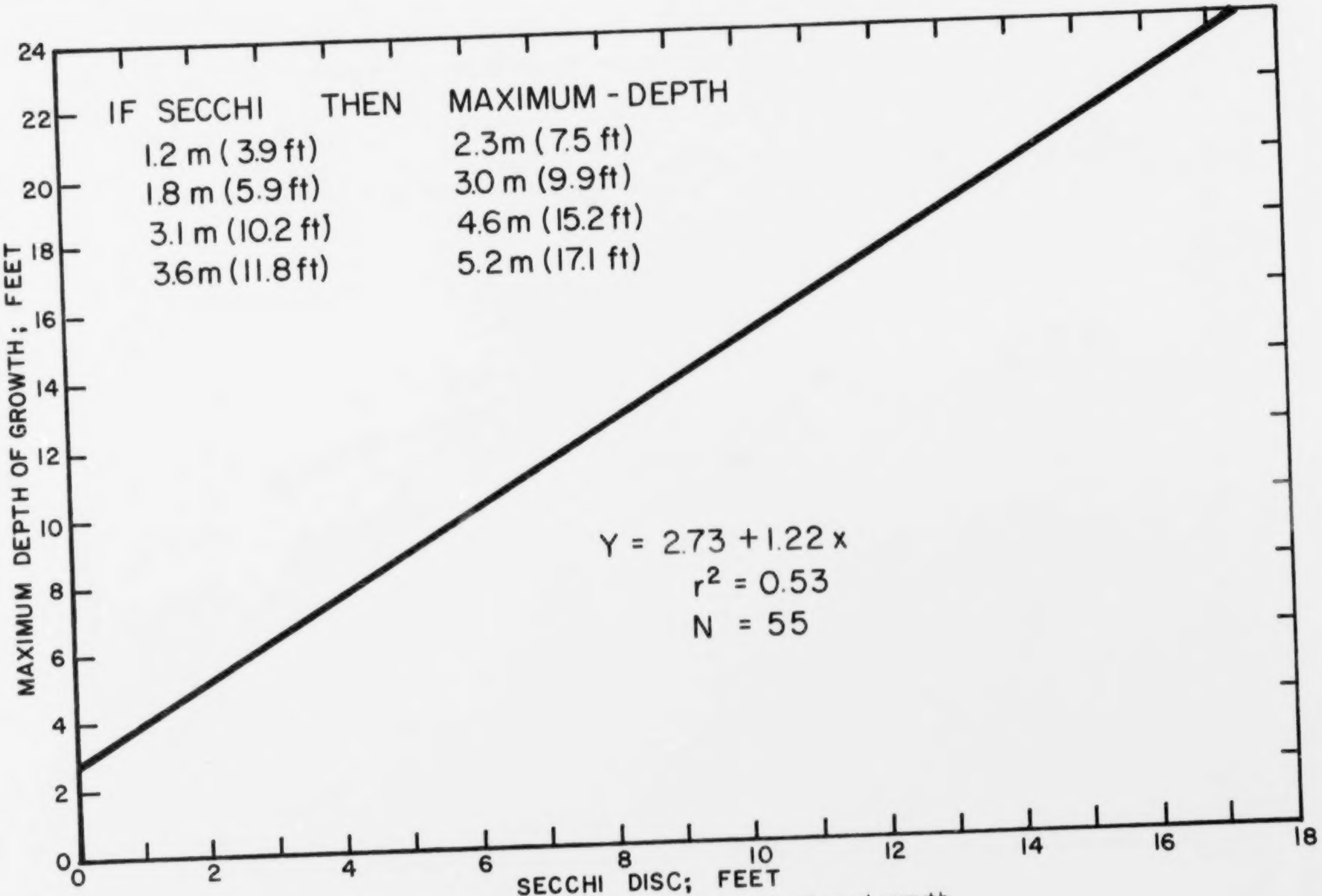


Figure 7. Water clarity versus depth of weed growth.

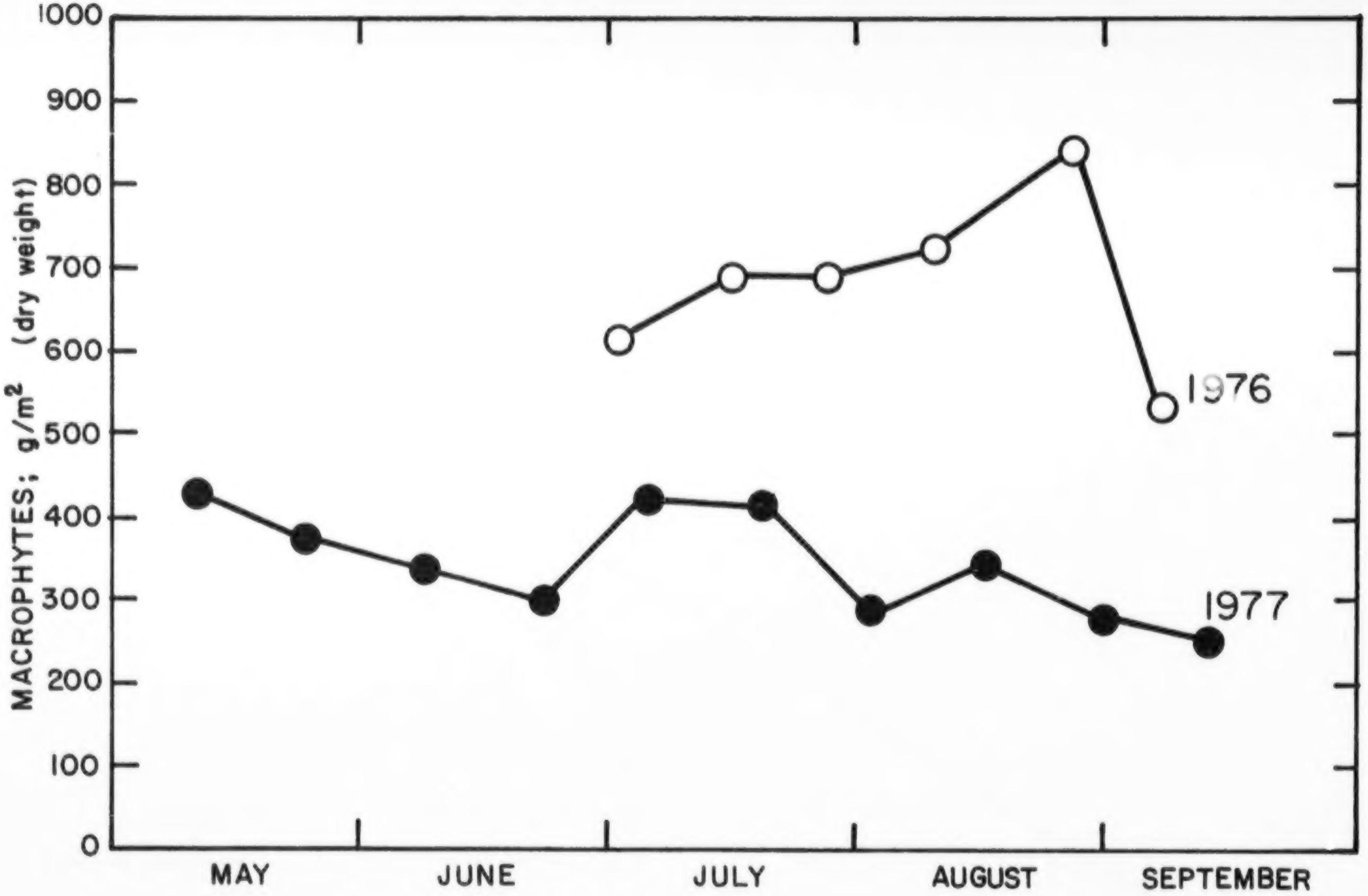


Figure 8. Trend in macrophyte biomass during 1976 and 1977.

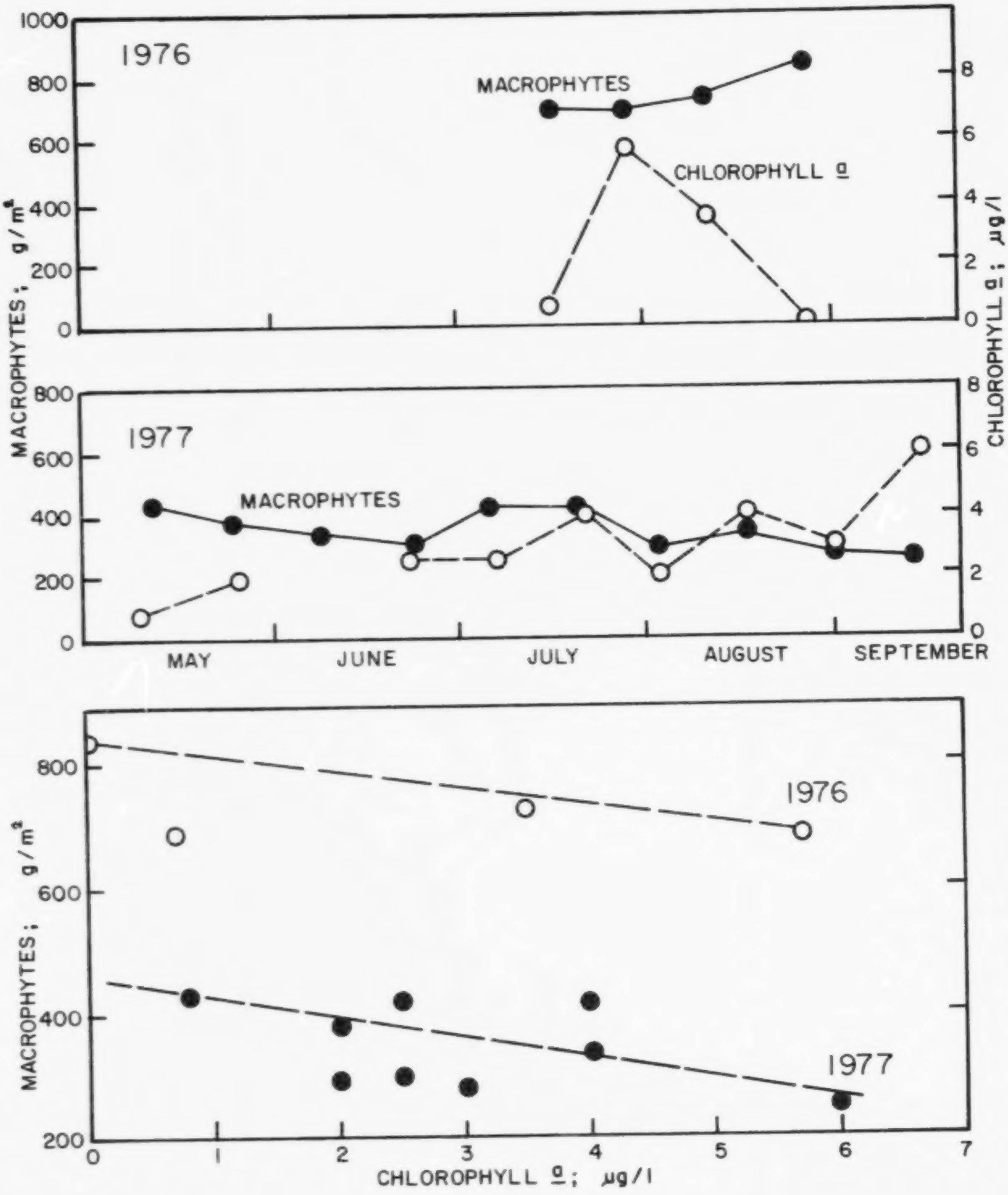


Figure 9. Comparison of chlorophyll a and macrophyte biomass.

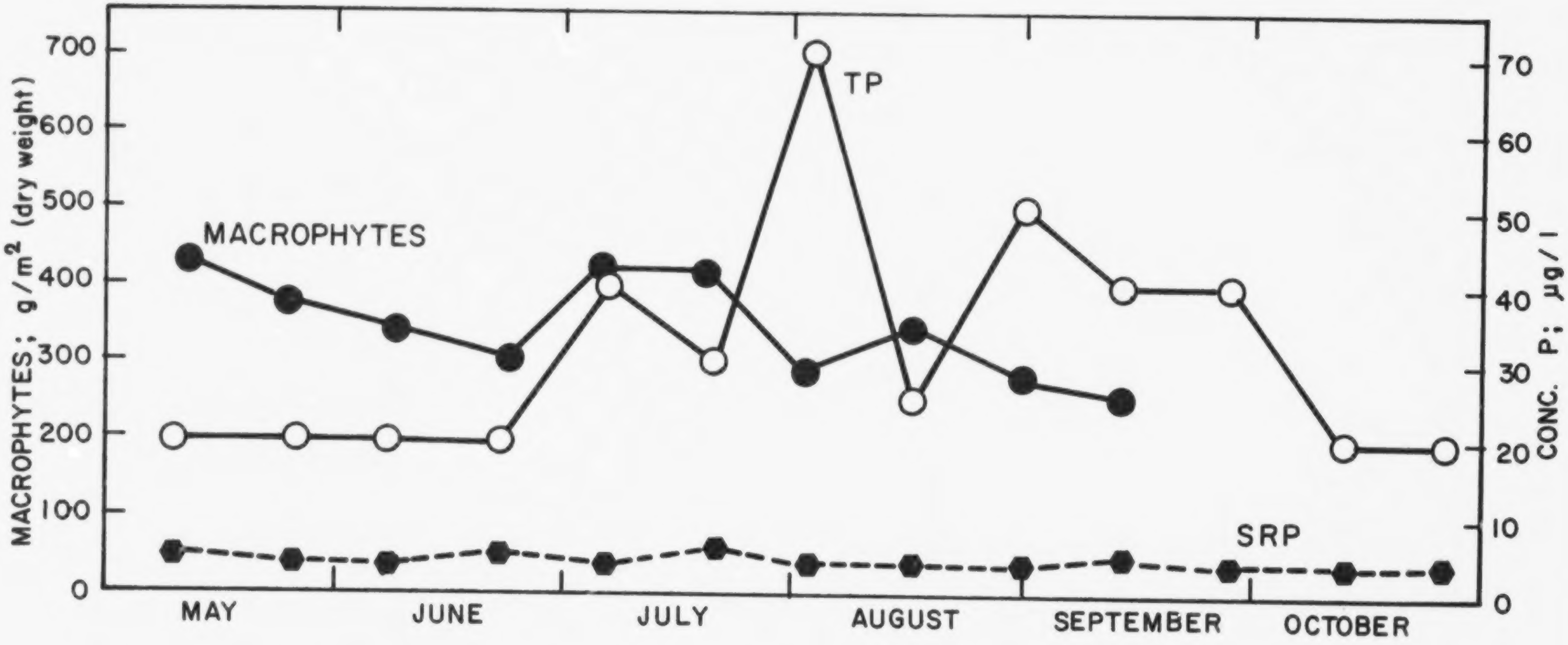


Figure 10. Macrophyte biomass and inlake P during 1977.

TABLE 4. COMPARISON OF MACROPHYTE FREQUENCY OCCURRENCE AT THE 0-0.5 m DEPTH FOR SOFT VERSUS HARD BOTTOM

Species	1976		1977	
	Soft	Hard	Soft	Hard
Potamogeton Robbinsii	94	61	98	79
Potamogeton amplifolius	23	4	28	14
Potamogeton praelongus	0	0	8	1
Potamogeton illinoensis	23	53	18	48
Megalodonta Beckii	42	1	52	5
Myriophyllum spicatum	41	7	16	4
Elodea canadensis	66	0	6	0
Myriophyllum exalbescens	25	0	3	0
Potamogeton pectinatus	35	23	16	29
Potamogeton zosteriformis	19	1	9	1
Heteranthera dubia	66	1	.1	0
Chara spp.	0	97	.1	78
Najas flexilis	17	23	4	18
Valisneria americana	.5	0	.1	0
Sagittaria spp.	0	0	.1	0
Number of samples	190	70	203	85

TABLE 5. COMPARISON OF MACROPHYTE FREQUENCY OCCURRENCE AT THE 0.5-1.5 m DEPTH FOR SOFT VERSUS HARD BOTTOM

Species	1976		1977	
	Soft	Hard	Soft	Hard
Potamogeton Robbinsii	98	45	98	91
Potamogeton amplifolius	44	55	41	49
Potamogeton praelongus	1	0	6	0
Potamogeton illinoensis	26	35	27	89
Megalodonta Beckii	42	2	58	7
Myriophyllum spicatum	33	2	6	1
Elodea canadensis	43	0	4	0
Myriophyllum exalbescens	12	3	2	0
Potamogeton pectinatus	29	12	16	9
Potamogeton zosteriformis	16	2	12	0
Heteranthera dubia	28	0	0	0
Chara spp.	0	90	0	8
Najas flexilis	8	50	1	13
Valisneria americana	.6	0	0	0
Sagittaria spp.	0	0	0	0
Number of samples	180	60	188	76

TABLE 6. COMPARISON OF MACROPHYTE BIOMASS BETWEEN 1976 AND 1977 FOR EACH BOTTOM TYPE*

	Soft Bottom		Hard Bottom	
Water Depth	1976	1977	1976	1977
0 - 0.5 m	790	345	38	76
0.5 - 1.5 m	898	432	71	125
>1.5 m	659	335	--	--

TABLE 7. COMPARISON OF MACROPHYTE BIOMASS BETWEEN WATER DEPTHS AND BETWEEN BOTTOM TYPES*

		0-0.5 m	0.5-1.5 m	>1.5 m
Potamogeton Robbinsii				
	Hard	25	55	---
	Soft	305	389	128
P. ampliforlius				
	Hard	3	22	---
	Soft	5.3	12	84.5
P. praelongus				
	Hard	0.2	0	---
	Soft	0.7	1.2	46.4
P. illinoensis				
	Hard	8.4	44	---
	Soft	3	4.7	6.5
Megalodonta Beckii				
	Hard	1.0	1.8	---
	Soft	8.1	17.5	0

* Biomss in g/m^2 dry weight.

Table 7, P. illinoensis, P. amplifolius and P. Robbinsii would be expected to dominate in the 1.5-3.0 meter hard bottom zone.

If water clarity is sufficient to permit macrophyte growth on the soft sediments at 3+ meters, the 4 species now dominant at 1.8 meters would be expected to invade the area (Table 7). P. amplifolius normally produces peak biomass at 1-3 meters but will grow to a depth of 5 meters. Little is known about the depth distribution of the other 3 species, but they are often found in deep water. Of the 5 most abundant species at present, only Megalodonta Beckii will undergo a sharp reduction after dredging.

FISH

In the spring length measurements were taken from all species and scale samples were collected from the primary species for age-growth analyses. The sport fishery is comparatively poor on this lake. The predominance of small, slow growing fish is thought to be due primarily to the extensive macrophyte beds. The original plan included calculation of growth rates prior to, during, and following treatment. The changes in growth could then be related to other environmental variables. The fish species present last spring are shown in Table 8. The extent of the fishkill has not yet been determined, but young-of-the-year have since been observed for several species.

INVERTEBRATES

Benthic invertebrate samples were collected with an Eckman dredge in April. Eight samples were taken; four over sand bottom and four over soft sediments. Additional sampling will be conducted this spring prior to the start of dredging in order to provide a solid base for comparison with during the post-dredging populations.

Zooplankton are being collected biweekly May through September. The work is non-intensive, involving a single tow from the bottom to the surface with a #20 mesh net near the center of the lake. These samples are being examined for species composition and enumeration and size distribution. Significant reduction in the population would be expected during the dredging activity if high turbidity occurs in the lake. And a whole new group of species should invade the subsequent lake (e.g. limnetic area).

SEDIMENTS

An important question surrounding any dredging project is the permanency of treatment. Using a piston corer near the lake center, 100 cm of sediment was removed, sliced into 2.5 cm segments, and dated by the Pb 210 method. The average infilling rate over the past 100 years has been 0.5 cm/yr. A technique has not yet been selected to date the deeper sediments.

An isopach map will be constructed by probing for the depth of soft sediments in the winter immediately following the dredging operation and in the last year of the study. The information will be used to evaluate whether the 10:1 slope, maximum depth of 6.1 meters can be maintained or whether the sediments will move toward and partially fill in the deeper area.

TABLE 8. FISH SPECIES PRESENT; MARCH 15, 1977

Species	Common Name
Lepomis macrochirus	Bluegill
Micropterus salmoides	Largemouth bass
Chaenobryttus gulosus	Warmouth
Pomoxis nitromaculatus	Black crappie
Ictalurus natalis	Yellow bullhead
Esox lucius	Northern pike
Perca flavescens	Yellow perch
Morone interrupta	Yellow bass
Lepomis cyanellus	Green sunfish
Ambloplites rupestris	Rock bass
Catostomus commersonnii	White sucker

TABLE 9. DISSOLVED OXYGEN DEPLETION RATES INSIDE SOD CHAMBERS—JULY 5-13, 1977*

Chamber		Depletion Rate (mg/l/day)
#1	T	0.84
	B	>1.14
#2	T	0.79
	B	>1.16
#3	T	1.0
	B	>1.04
#4	T	0.96
	B	<0.98

* T = 10 cm (4 inches) from the top of the chamber.
B = 43 cm (17 inches) from the top of the chamber.

TABLE 10. WATER CHEMISTRY OF THE GROUNDWATER; MARCH/AUGUST, 1977

Parameter GW Well No.	1	2	3	4
SRP (μg/l)	<4	5	<4	221
TSP (μg/l)	<20	<20	<20	241
Chloride (mg/l)	27	17	29	29
Conductivity (microhmos/cm at 25¼C)	890	539	912	861
Nitrite/Nitrate-N (mg/l)	<.02	<.02	<.02	<.02
Ammonia-N (mg/l)	0.74	0.12	1.2	6.3
Total Organic-N (mg/l)	0.4	<0.2	0.7	2.6

Four opaque sediment oxygen demand (SOD) chambers are being used to determine the oxygen demand and the N and P release rates from the sediments. The chambers were installed near mid-lake during July 1977. Sampling was conducted 2-4 times per month from 10 and 43 cm below the top of the chamber. The chamber top is 61 cm above the sediment surface. Each chamber covers 1.17 square meters of lake bottom and contains 0.71 cubic meters of water. The rates will be determined prior to dredging, prior to and shortly after aluminum sulfate application, and in the last year of the project. This will permit determination from existing and newly exposed sediments, plus the effects of an overlying aluminum floc.

The sediments are exerting a significant demand on the oxygen resources of the lake (Table 9). Low levels were, however, never measured during summer, and the demand is apparently readily offset by photosynthetic activity and atmospheric oxygen transfer into the lake. Even during the winter, under ice cover, photosynthetic activity appears to be important (Figure 11). In January dissolved oxygen concentrations dropped rapidly and a severe fishkill resulted in early February. Warm weather occurred soon after, causing the disappearance of the overlying snow. Improved light penetration apparently enabled the increase in dissolved oxygen levels through the remainder of the winter.

The highest concentration of SRP in the SOD chambers was 19 µg/l (Figure 12) under anoxic conditions. In each chamber the concentration appeared to reach a peak and then decline to a slightly lower level. Under aerobic conditions within the lake, the highest value was 6 µg/l (Figure 10). Ammonia-N levels continued a steady increase through the period of observation. The concentration rose 0.05-0.07 mg/l/day, producing a release rate of about 30-42 mg/m^2 of lake bottom/day. The dissolved oxygen values represented for SOD #1 were typical for each of the 4 chambers. Sediment chemistries to be conducted later this spring will include total and interstitial P and N analyses.

HYDROGEOLOGY

The present investigation is concerned with determining the flow pattern and water budget in the vicinity of the lake. Data are provided by a network of 20 observation wells surrounding the lake (Figure 13). Other data have been obtained through the use of seepage meters implanted in the lake bottom (Figure 14). Fluctuations in lake level have also been recorded. In addition, a piezometer nest consisting of three wells has been emplaced in the center of the lake through the winter ice. This nest will be removed before the spring thaw.

To data, eight observation wells have been providing data since July, 1976. These eight wells consist of four pairs placed at approximately the NE, SE, SW, and NW extremes of the lake. Each pair consists of one well within a few feet of the lake with the other well placed directly away from the lake at a distance of roughly 100 feet. The purpose of these pairs is to provide nearshore groundwater gradients. An additional nine wells have been under observation since November, 1977. Their purpose is to fill out the water table profile around the lake. In January, 1978, three new wells were installed; two on the two hilltops around the lake to determine whether there is

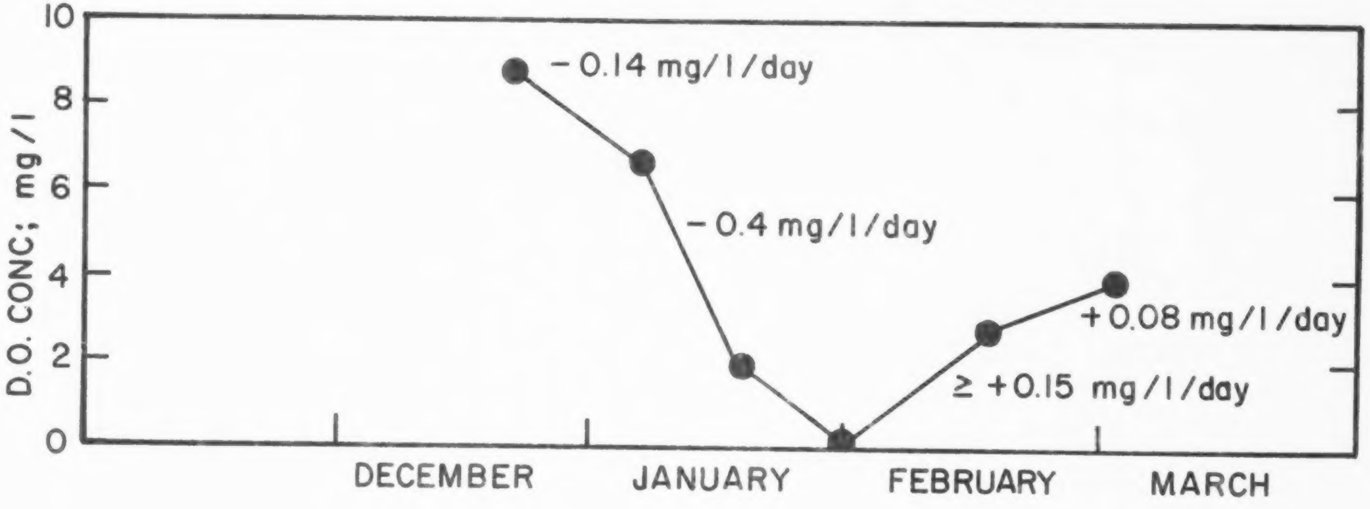


Figure 11. Inlake dissolved oxygen levels during 1976-77.

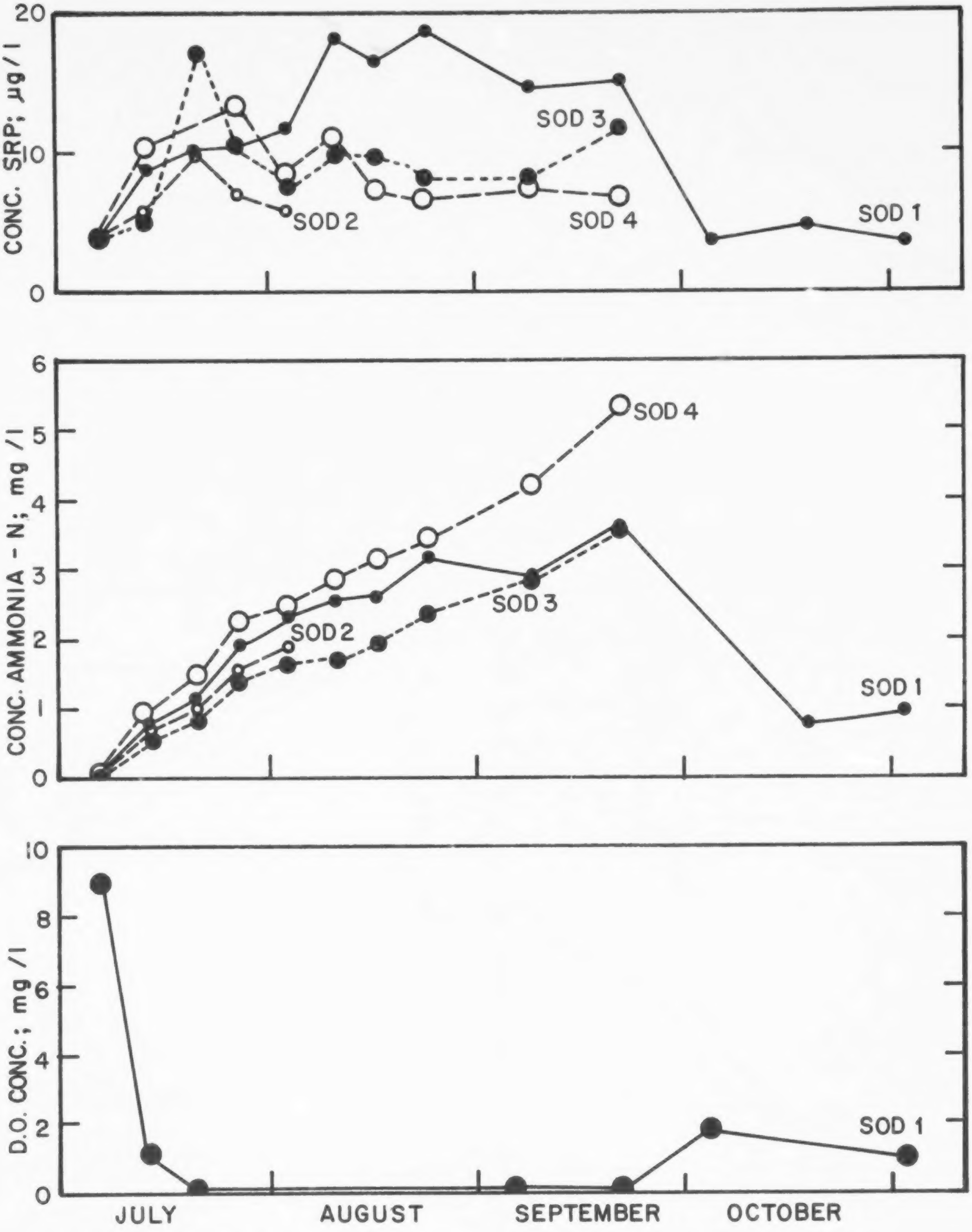


Figure 12. Release of P and N from sediments inside the SOD chambers (average concentration within chamber).

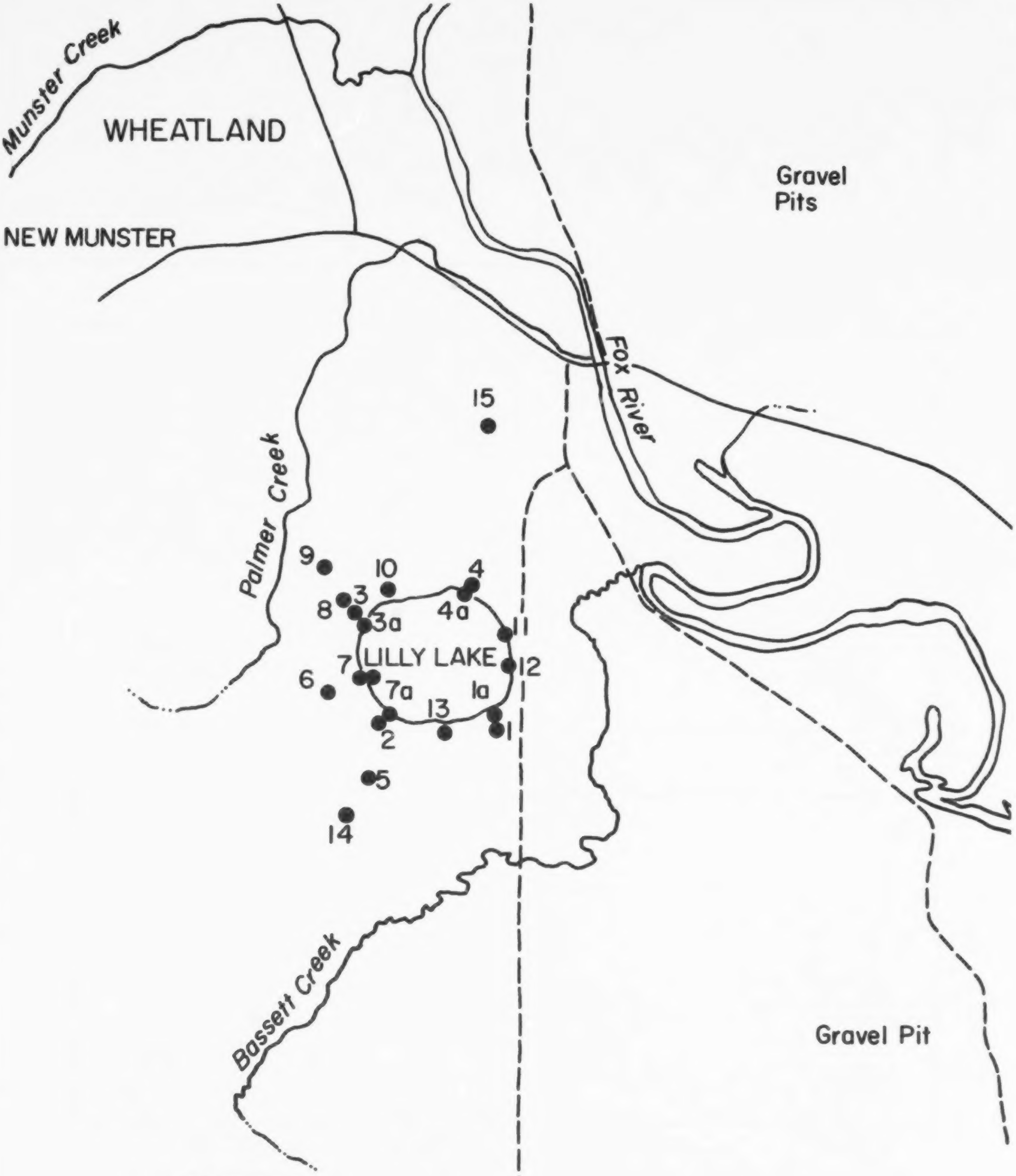
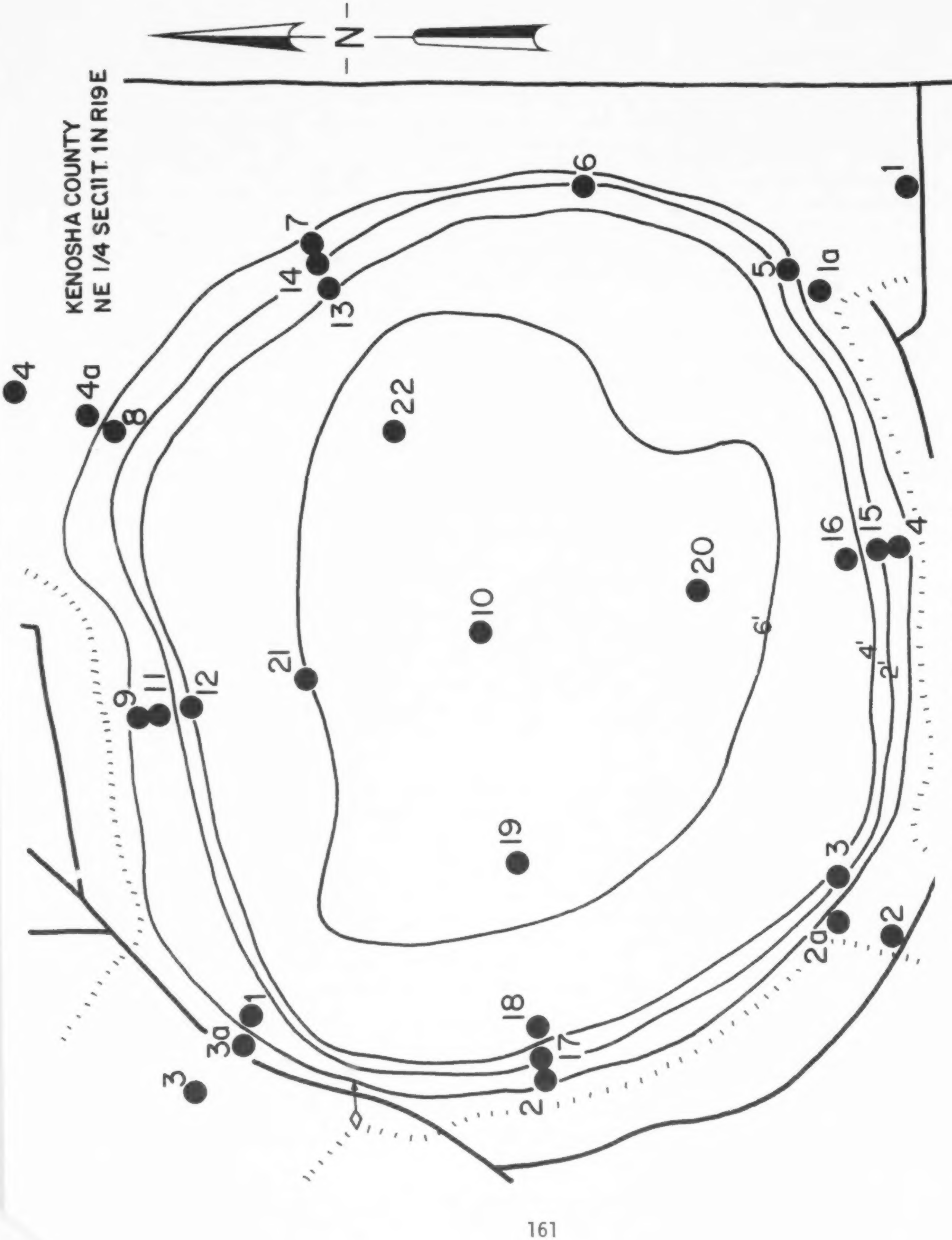

Munster Creek
WHEATLAND
Gravel Pits
NEW MUNSTER
Fox River
15
Palmer Creek
9
10
4
8
3
3a
4a
11
LILLY LAKE
7
12
6
7a
1a
13
2
1
5
14
Bassett Creek
Gravel Pit
BASSETT

Figure 13.


KENOSHA COUNTY
NE 1/4 SEC.11 T. 1N R19E
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6'
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2'

Figure 14. Location of inlake seepage meters.

significant groundwater mounding under those hills, and one to replace a well which we will lose access to in the spring.

Lake level and water levels in the original four pairs of observation wells have been plotted against time. These plots provide both a rough picture of water table gradients at four points along the lake shore, and an indication of how those gradients vary with time. They indicate clearly that groundwater is entering the lake in the SW (wells 2 and 2a), lake water is entering the groundwater system in the SE (wells 1 and 1a) and NE (wells 4 and 4a), and flow sometimes occurs both ways in the NW (wells 3 and 3a) but usually from lake to groundwater system.

Data from the other wells confirm the hypothesis that there is indeed a gradient from the SW (well 5) and WSW (wells 6 and 7) which results in movements of groundwater into the lake. When wells 8 and 9 are taken into accounts with wells 3 and 3a in the NW, there is a very clear gradient away from the lake, that is lake water is entering the groundwater system there. In the spring of 1977 however, there was a period when it appeared that groundwater was entering the lake at that point. This could possibly be related to the spring melt. Although lake levels have not been obtained since the lake froze in November, 1977, water levels in well 13 in the S and in wells 11 and 12 in the E have been consistently below the mean lake level, indicating probable discharge of lake water into the groundwater system.

Well 14 was installed in January, 1978, on a hill to the SW of the lake. It extends down to about 25 feet below the lake surface. The few readings taken so far in that well indicate a water level slightly higher than that of the lake, although no significant mounding of water seem to occur there. Well 15 was also installed in January, 1978, on a hill to the N of the lake. A barrier which could not be penetrated by our drilling mechanism was encountered about 25 feet above the lake surface, and the hole is dry. Information from commercial drillers, however, indicates that the water table beneath that hill is below lake level.

Beginning in May, 1977, and continuing until October, 1977, nine seepage meters, as described by Lee (1977), were implanted in the nearshore lake bottom sediments around the perimeter of the lake. In July, 1977, an additional thirteen seepage meters were installed in two lines crossing the lake, roughly N-S and E-W, and were monitored through September, 1977. Plastic bags containing 500-800 ml of water, and having a total capacity of about 1500 ml, were affixed to each seepage meter. After four to six hours, the plastic bags were removed and the change in water volume was measured. An increase in water volume indicated an influx of groundwater into the lake, while a decrease in water volume indicated that water was leaving the lake.

The data obtained from these seepage meters are in overall agreement with the flow picture derived from the observation wells. They clearly show that groundwater is entering the lake on the west side and leaving on the east and north, with the south side showing both groundwater inflow and outflow. This is in agreement with the regional flow system as delineated by Cotter, et al. (1969), which shows a generally easterly movement of groundwater towards the Fox River.

The distribution of seepage magnitude however does not coincide with the model constructed by McBride and Pfannkuch (1975). Their model indicates that the majority of seepage should occur in a narrow band near the lake shore, with an exponential decrease in seepage with increasing distance from shore. Our experience at Lilly Lake has been that while in general the larger magnitude seepages do seem to occur near shore, there is not a regular exponential decrease with increasing distance from shore, and in fact, relatively high magnitude inflows were consistently recorded in the very center of the lake.

An explanation for this seeming anomaly might be that there are zones of relatively high permeability beneath the lake which cause deep springs to occur in the vicinities of some of our seepage meters. Such a situation has been reported by Cooke and McComas (1974) at West Twin Lake in Ohio. We have attempted to confirm the strong inflows in the center of the lake by means of a three-well piezometer nest emplaced through the winter ice, but to date, the deepest well has not equilibrated sufficiently for reliable measurements to be taken, and information from the two shallower wells is ambiguous.

At this point it is possible to draw several conclusions regarding the probable effects of the dredging project on Lilly Lake. It appears that the organic sediments to be dredged have little effect on the flow system. They are extremely permeable, and that portion to be dredged, at least, does not act to seal the lake off from the groundwater system. It is expected that when dredging begins, lake level will decline, all outgoing gradients around the lake will reverse, and groundwater will flow into the lake around its entire perimeter. As long as this condition persists, we expect that the relatively high level of nutrients (Table 10) found in the groundwater system at the NE extreme of the lake will flow into the lake, rather than away from it. When dredging is completed however, and equilibrium reestablished, the NE sector should revert to its original state of outflow.

The dredged sediments are to be disposed of by piping them to an inactive gravel pit situated just west of the Fox River (Figure 1). It is thought that the high water content of the sediments will allow them to be transported as a slurry, with no additional water needed. The gravel pit will have an earthen dike constructed across its open end and the sediments will simply fill the pit. Considering the high permeabilities of sand and gravel, however, it is thought that the water within the sediments will tend to seep away, most probably in the direction of the nearby Fox River. A network of observation wells will be installed around the gravel pit, and between the pit and the river, to try to follow this anticipated flow. In the event that the amount of sediment dredged exceeds the capacity of the gravel pit, the remainder will be used in an experimental program of spray irrigation on nearby farm lands.

All of our observation well and seepage meter monitoring will continue, insofar as possible, through the dredging period and until the system has reestablished equilibrium. The response of the system to the dredging will then be evaluated in the hope of developing a computer model of the process applicable to other similar situations. If the Lilly Lake project is as successful as we anticipate it will be, we hope to set a precedent for many future lake renewal projects.

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EVALUATION OF DREDGING AS A LAKE RESTORATION TECHNIQUE, LAKE LANSING, MICHIGAN

by

C. D. McNabb*

Lake Lansing in Ingham County, Michigan, has been selected by the U.S. Environmental Protection Agency as a site to test the efficacy of hydraulic dredging as a lake restoration technique. This lake is the only major surface water resource for recreation in the Lansing metropolitan region. It has become gradually eutrophic with intensive use over the past several decades. Blue-green algae blooms and extensive beds of vascular hydrophytes have caused the economic and aesthetic value of the resource to decline.

Wastes of the residential and business community in the watershed of the lake were sewered nearly a decade ago to retard nutrient input. The watershed is presently stabilized in terms of erosion. Proposed dredging will deepen the littoral zone of the lake and improve beach and public park shorelines. The objectives of this study are to (1) quantify the annual nutrient budget for the lake basin, (2) correlate changes in production of macrophytes, algae, invertebrate animals and fish with physical and chemical changes that occur in the lakes as a result of dredging, and (3) to evaluate the ecological impact of dredged materials on disposal sites. Evaluation of dredging will extend over a period of five or more years. Intensive sampling will begin in April of 1978.

LIMNOLOGICAL STUDIES

The research design for limnological portions of the study has been based on certain physical features of the Lake Lansing basin. Marsh and Borton (1974) have identified the features of drainage in the watershed as shown in Figure 1. In addition to precipitation on the surface, water enters the lake via springs and seepage through surrounding marsh lands. Young et al. (1974) calculated a mean annual discharge of 0.1 m^3 sec^{-1} and estimated the nominal retention time of Lake Lansing to be 1.6 years at present. For purposes of sampling and data interpretation, the water in the lake is virtually standing, with surface winds, rather than stream-flow, being responsible for the mixing that occurs.

A bathymetric map of the present state of the lake basin is presented in Figure 2. It has been taken from the Lake Inventory map of the Michigan

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INGHAM COUNTY, MICHIGAN
T 4N R 1W

Watershed Area - 8.42 sq. km.
Watershed Perimeter - 14.32 km.
Watershed Shape Factor - 1.39
Lake Area - 182 ha.
Lake Perimeter - 5.88 km.
Lake Shape Factor - 1.22
Lake Area : Watershed Area - 1 : 4.58

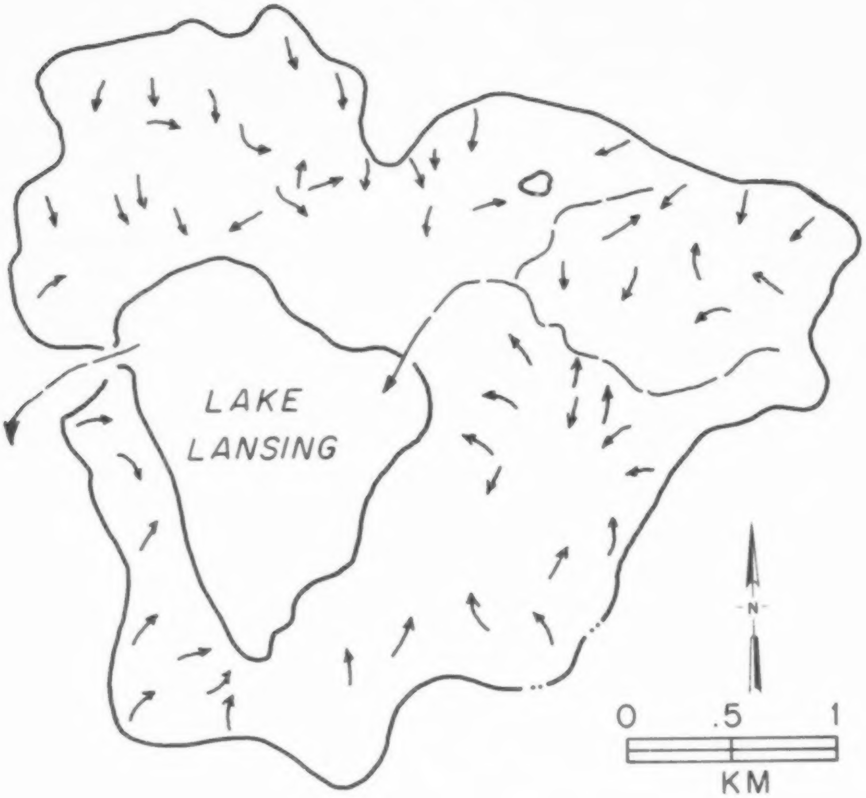
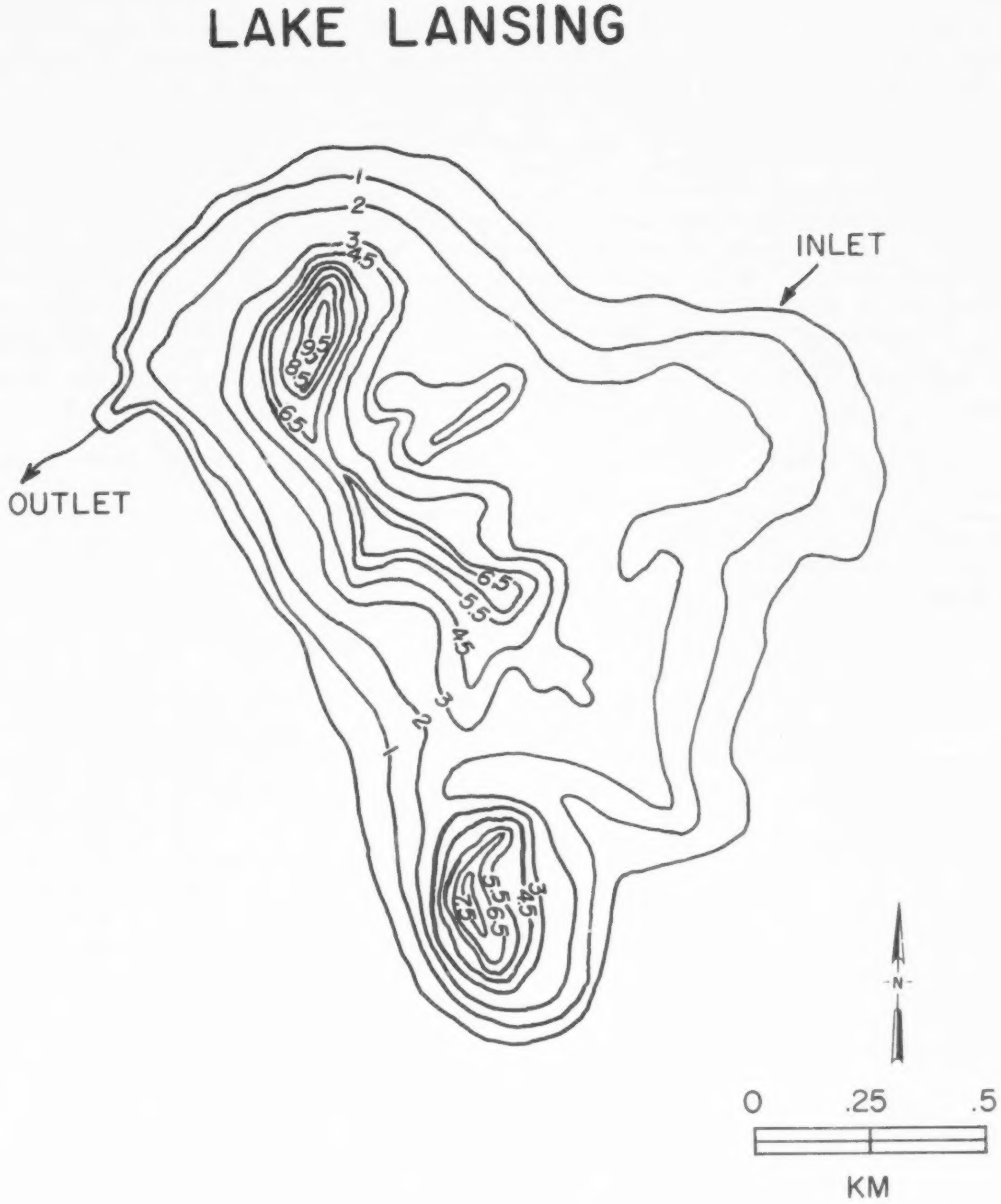


Figure 1. The pattern of overland flow in the Lake Lansing watershed and statistics of size and shape of the watershed and lake. Shape factors are the ratio of length of the perimeter to the length of the perimeter of a circle having the enclosed area (from Marsh and Borton, 1974).


LAKE LANSING
1
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3
4.5
5.5
6.5
7.5
INLET
OUTLET
N
0
.25
.5
KM

Figure 2. Bathymetric map of Lake Lansing. Depth 4.5 m is the present depth to which macrophytes of the littoral grow.

Institute for Fisheries Research dated 8/14/38, and modified according to depths at locations sampled in recent years by limnology classes at Michigan State University. A north and south basin exist in the lake as shown.

Figures 3, 4, 5 and 6 were obtained from the bathymetric map using a polar compensating planimeter. These figures accentuate the shallowness of the present basin. The littoral zone of the lake, as defined by the depth to which rooted aquatic macrophytes grew in 1974 and 1975, extends to the 4.5 m contour. This is the approximate depth of the suface of the hypolimnion in the south and the north basin in summer (Young et al., 1974). It can be seen from Figures 3 and 4 that the combined area of the hypolimnetic surfaces over the two deep holes in the lake is 15% of the total lake surface. The truly open-water, pelagial regions of the lake are thereby confined to 15% of the surface area. The littoral region lies beneath 85% of the surface area of the lake.

Figures 5 and 6 show that only 1.73% of the volume of the lake is contained in the hypolimnion of the south basin, and only 6.6% of the volume is contained in the hypolimnetic region of the north basin. Nearly all of the existing limnological data from the lake has been previously obtained by sampling over and into the two deep holes (cf. Young et al., 1974). The preponderance of area and volume in the littoral region, as well as the lack of limnological data gathered for that region, have been important factors in causing sampling emphasis in this document to be placed in the shallows.

The littoral zone of Lake Lansing is well occupied by a diverse native flora. Emergent cattails, bulrushes, and species of Sagittaria are inshore of water lilies along undeveloped portions of the shoreline. An Elodea canadensis-Najas flexilis association dominates the submersed macrophyte community of the south basin. Characean meadows and mixed communities of potamogetons, water milfoils and Vallisneria americana are the dominant associations of the north basin. Epipelic algae, principally blue-greens, commonly begin development on the littoral sediments and rise to float freely at the surface in summer. Filamentous algae of the metaphyton are relatively abundant. A succession of algae in the plankton occurs through spring and summer, culminating in blue-greens. Littoral and phytoplanktonic production has never been quantified for Lake Lansing (except for the effort of Young et al., 1974, using meager CO_2 data). Young et al. (ibid) found Secchi disc transparency in the growing season to be in the range of 1-3 m.

McNabb (1975) has suggested that recreational lakes of southern Michigan can be positioned within Figure 7 by analyses of standing crops of the various plant associations. Gathering the quantitative data to locate Lake Lansing within this scheme before and after dredging has been given a major emphasis in the limnological portion of this proposal. Visual observations on the lake in recent years would suggest that the crops of planktonic and filamentous algae, and macrophytes would presently place the lake on the right-hand portion of the "plateau of nutrient competition."

Figure 8 (ibid) suggests the impact of two exotic species on the native submersed flora of southern Michigan lakes. Myriophyllum spicatum (eurasian water milfoil), and Potamogeton crispus (curly-leafed pondweed), both intro-

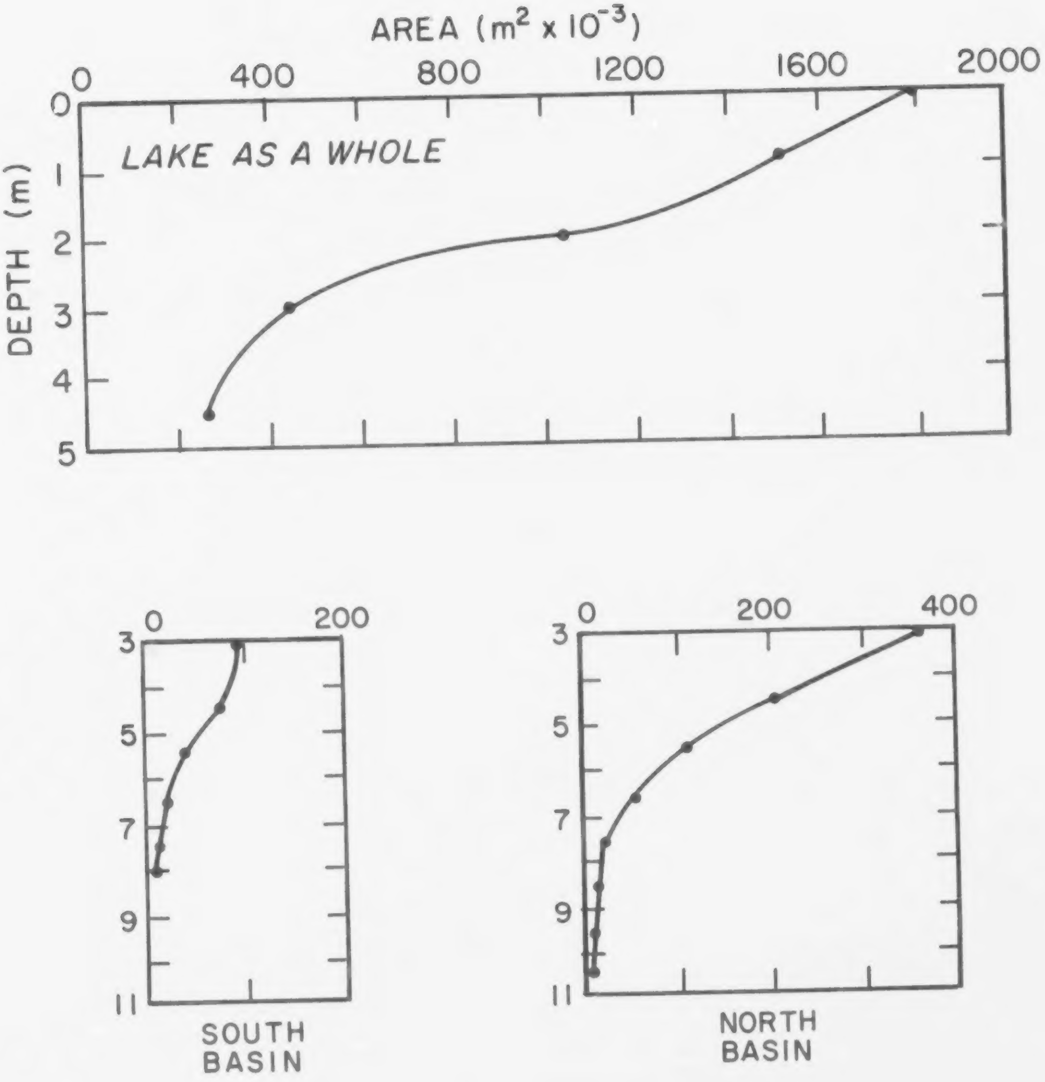


Figure 3. Depth-area curves for the littoral of Lake Lansing above and for the two deep holes of the lake below.

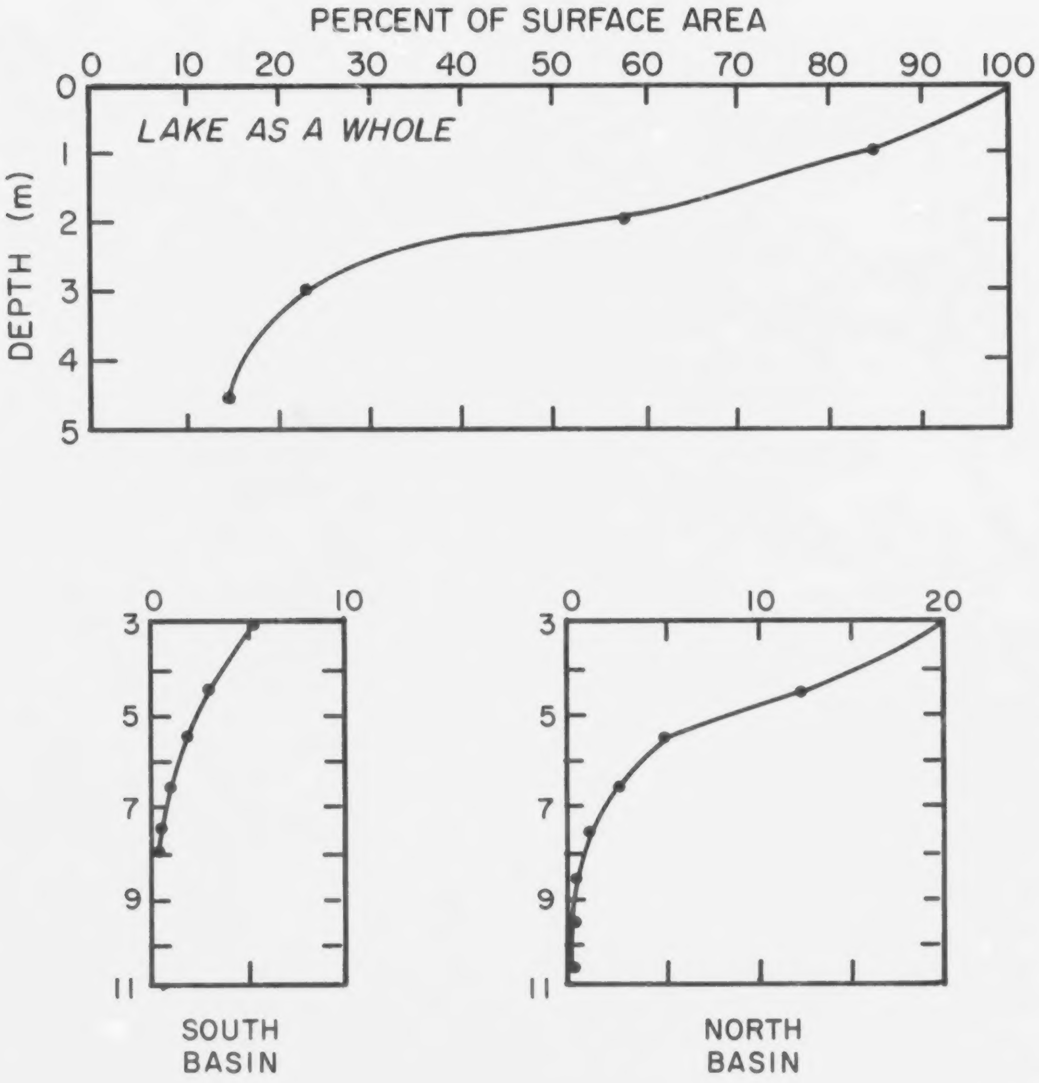
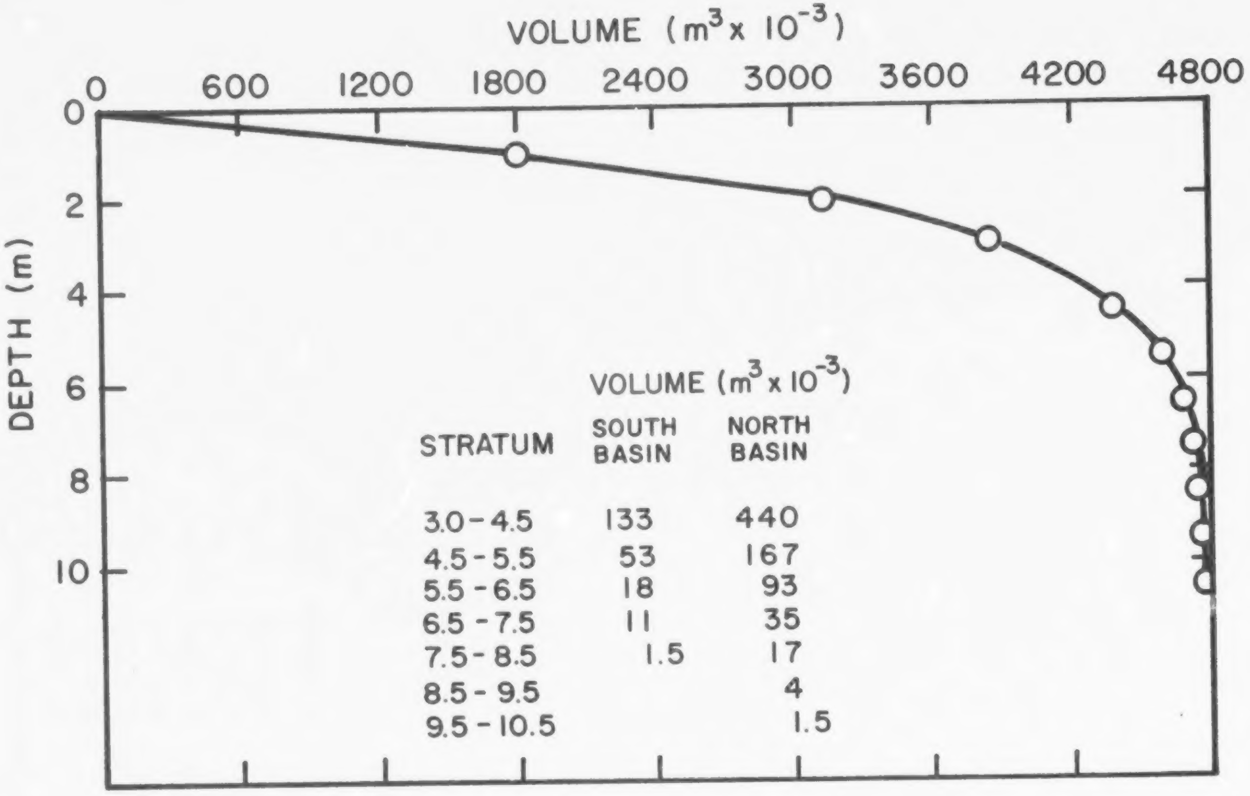
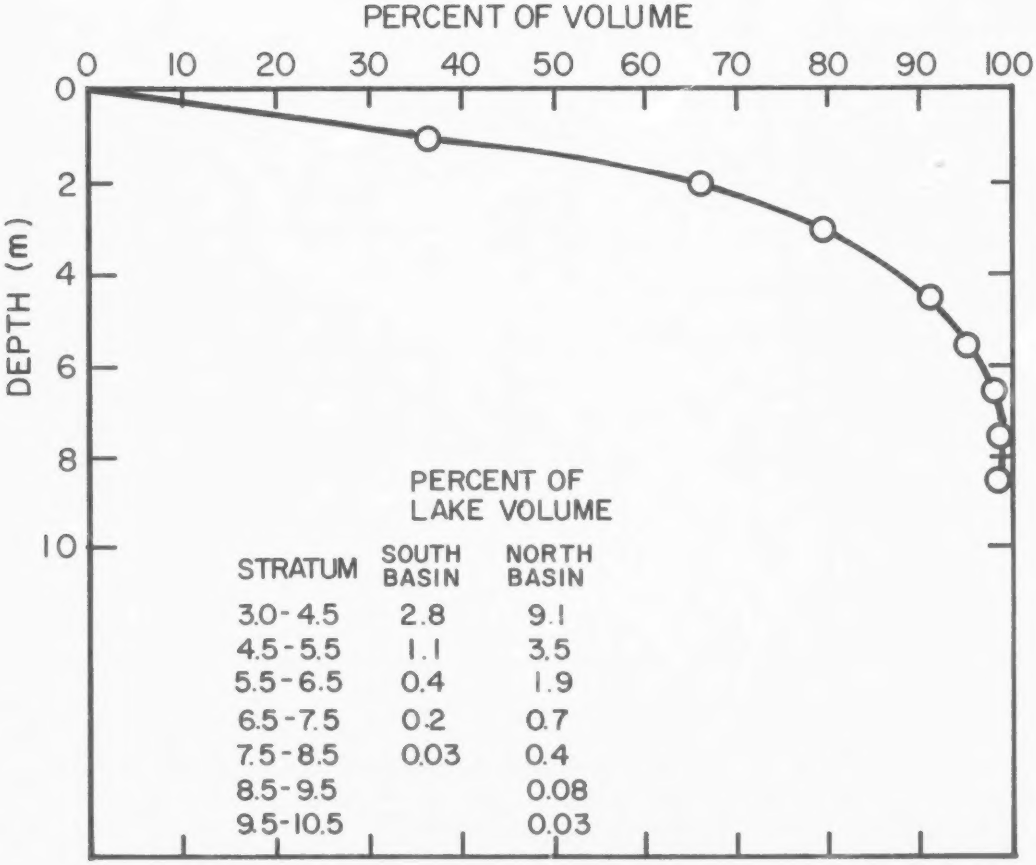


Figure 4. Depth-percent surface area curves for the littoral of Lake Lansing above and for the two deep holes of the lake below.



STRATUM	VOLUME ($m^3 \times 10^{-3}$) SOUTH BASIN	NORTH BASIN
3.0 - 4.5	133	440
4.5 - 5.5	53	167
5.5 - 6.5	18	93
6.5 - 7.5	11	35
7.5 - 8.5	1.5	17
8.5 - 9.5		4
9.5 - 10.5		1.5

Figure 5. Depth-volume curve for Lake Lansing with tabled volumes for strata in the two deep holes in the lake.



PERCENT OF LAKE VOLUME

STRATUM	SOUTH BASIN	NORTH BASIN
3.0-4.5	2.8	9.1
4.5-5.5	1.1	3.5
5.5-6.5	0.4	1.9
6.5-7.5	0.2	0.7
7.5-8.5	0.03	0.4
8.5-9.5		0.08
9.5-10.5		0.03

Figure 6. Depth-percent volume curve for Lake Lansing with tabled percentages of total volume for strata in the two deep holes of the lake.

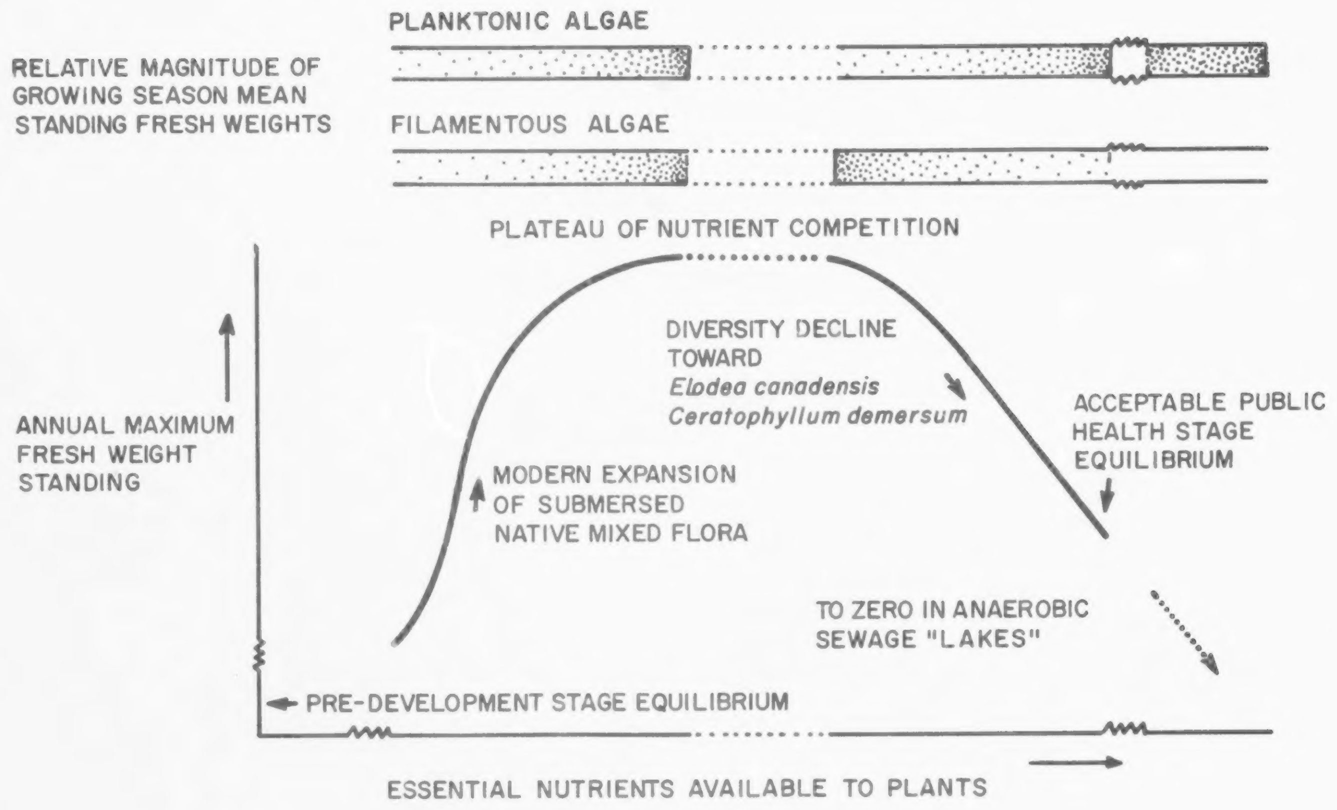

PLANKTONIC ALGAE
RELATIVE MAGNITUDE OF GROWING SEASON MEAN STANDING FRESH WEIGHTS
FILAMENTOUS ALGAE
PLATEAU OF NUTRIENT COMPETITION
DIVERSITY DECLINE TOWARD *Elodea canadensis* *Ceratophyllum demersum*
ACCEPTABLE PUBLIC HEALTH STAGE EQUILIBRIUM
ANNUAL MAXIMUM FRESH WEIGHT STANDING
MODERN EXPANSION OF SUBMERSED NATIVE MIXED FLORA
TO ZERO IN ANAEROBIC SEWAGE "LAKES"
PRE-DEVELOPMENT STAGE EQUILIBRIUM
ESSENTIAL NUTRIENTS AVAILABLE TO PLANTS

Figure 7. Generalized relationship of standing crops of aquatic plants in recreational lakes of southern Michigan of increasing fertility (McNabb, 1975).

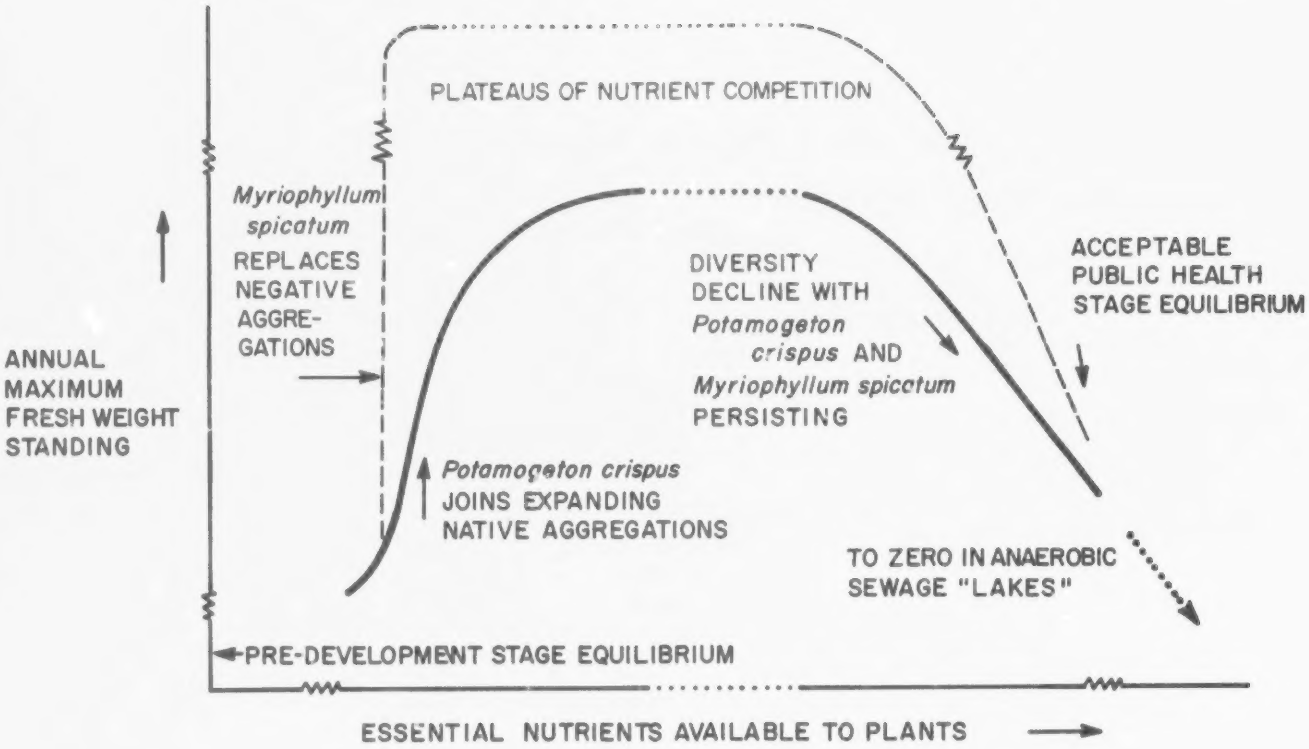


Figure 8. Generalized relationship of standing crops of eurasian water milfoil (*M. spicatum*) and standing crops of native submersed macrophytes and *P. crispus* in recreational lakes of southern Michigan of increasing fertility (McNabb, 1975).

duced from Europe, are major weed species of the region. Both can exist in relatively nutrient poor lakes reaching high biomass by presumably tapping nutrient resources of the sediments. While the native flora and *P. crispus* do not cause severe weed problems in depths greater than 2 m, *M. spicatum* does reach the surface with high biomass from 4 m (Coffey and McNabb, 1974). As a weed, the latter plant thus removes more surface acres from recreational use than *P. crispus* or the native flora, and achieves a higher annual biomass for the lake as a whole as shown in the figure.

Neither of the exotic weeds is important in the flora of Lake Lansing at the present time. There is no reason at the present time to suspect that the habitat will be unsuitable for their growth in the post-dredging era. Dredging the littoral zone to 4 m (Snell, 1975) will not reduce its area from that presently existing. Post-dredging analyses of the littoral vegetation will be important in an evaluation of the technique as a restoration tool. Quantitative base-line data will be collected in the 1978 growing season.

Limnological theory has advanced to the stage of predicting that changes in the primary productivity of Lake Lansing will be causally related to changes in chemical and physical attributes of the environment. A comparison of the analyses of these attributes before and after dredging is a principal feature of this research. In particular, Wetzel (1975) has suggested that the rate of decalcification of the littoral-epilimnetic regions in a growing season is related to productivity so as to be more rapid in more productive systems. The high rate of decalcification in eutrophic systems in the absence of changes of the same order of magnitude in the more conservative Mg^{++}, Na^{+} and K^{+} can be at least partially implicated in lowering the monovalent:divalent cation ratio in the zone of production.

If plant productivity in the lake is depressed as a result of dredging, the rate of decalcification in the surface waters in summer should decrease, and the monovalent:bivalent cation ratio should decrease as well. A low ratio is typical of unproductive marl lakes in southern Michigan. This observation is relevant to the study for the reason that Lake Lansing appears to have progressed historically from a marl lake system to a eutrophic system with cultural development of the watershed. Figure 9 suggests that a depression of primary productivity as a result of dredging should cause the metabolism of Lake Lansing to shift in some degree away from its present eutrophic state toward that of either a marl lake of an oligotrophic lake. Changes in the cation ratio can be used as an index of the degree and the direction of this shift. For these reasons, analyses for the cations have been included in the proposal.

Analysis of changes in dissolved organic matter (DOM) are proposed here for the reasons shown by Figure 9. Fractions of this component in the water serve as chelators for essential metals (e.g., iron) and as an energy source for heterotrophic bacteria. Bacterial metabolism provides organic micronutrients (e.g., algal vitamins) and free-CO_2 to populations of primary producers. The rationale for including analyses for forms of nitrogen and phosphorus has been well discussed in the limnological literature, and is illustrated in Figure 9 as well. Silicon has been included in the study because of the relationships between it and diatom productivity established by Schelske and

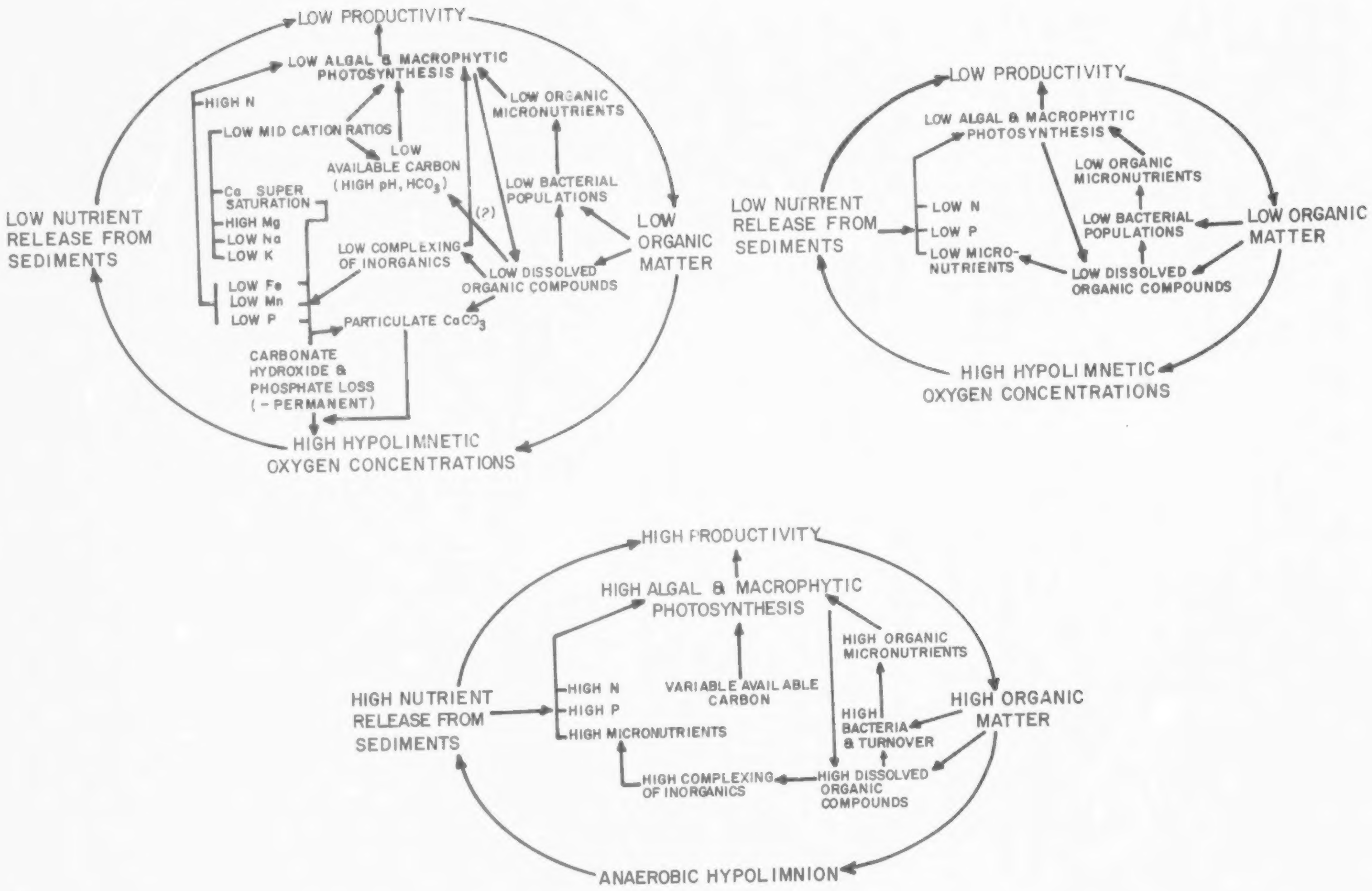

LOW PRODUCTIVITY
LOW ALGAL & MACROPHYTIC PHOTOSYNTHESIS
HIGH N
LOW MID CATION RATIOS
LOW AVAILABLE CARBON (HIGH pH, HCO3)
LOW ORGANIC MICRONUTRIENTS
Ca SUPER SATURATION
HIGH Mg
LOW Na
LOW K
LOW Fe
LOW Mn
LOW P
(?)
LOW BACTERIAL POPULATIONS
LOW COMPLEXING OF INORGANICS
LOW DISSOLVED ORGANIC COMPOUNDS
PARTICULATE CaCO3
CARBONATE HYDROXIDE & PHOSPHATE LOSS (- PERMANENT)
LOW NUTRIENT RELEASE FROM SEDIMENTS
LOW ORGANIC MATTER
HIGH HYPOLIMNETIC OXYGEN CONCENTRATIONS
LOW PRODUCTIVITY
LOW ALGAL & MACROPHYTIC PHOTOSYNTHESIS
LOW ORGANIC MICRONUTRIENTS
LOW N
LOW P
LOW MICRO-NUTRIENTS
LOW BACTERIAL POPULATIONS
LOW DISSOLVED ORGANIC COMPOUNDS
LOW NUTRIENT RELEASE FROM SEDIMENTS
LOW ORGANIC MATTER
HIGH HYPOLIMNETIC OXYGEN CONCENTRATIONS
HIGH PRODUCTIVITY
HIGH ALGAL & MACROPHYTIC PHOTOSYNTHESIS
HIGH ORGANIC MICRONUTRIENTS
HIGH N
HIGH P
HIGH MICRONUTRIENTS
VARIABLE AVAILABLE CARBON
HIGH BACTERIA & TURNOVER
HIGH COMPLEXING OF INORGANICS
HIGH DISSOLVED ORGANIC COMPOUNDS
HIGH NUTRIENT RELEASE FROM SEDIMENTS
HIGH ORGANIC MATTER
ANAEROBIC HYPOLIMNION

Figure 9. Limnological relationships that have served as a guide for selecting the measurements to be made in the Lake Lansing study (from Wetzel, 1975).

Stoermer (1972). King (1972) has shown that the concentrations of free-CO_2, determined knowing pH, alkalinity and temperature, is at least partially implicated in favoring obnoxious blue-greens at low concentration.

European and American workers have used the rates of oxygen depletion or carbon dioxide increase in the hypolimnion in summer or the tropholytic zone in winter as indices of rates of production under low input of allochthonous organic materials to lakes (cf. Wetzel, 1975). While Young et al. (1974) have summarized useful data concerning these parameters, measurements of each are proposed here for a short pre-dredging interval in 1978 to strenghten the base of data. Each of these analyses is to be done in the post-treatment period and these data will be related to changes in primary productivity.

It is proposed that during the period of the study, samples of the lake and its outflow be analyzed for arsenic, copper and mercury on a regular schedule. The first two of these have been used over the years to control aquatic weeds in Lake Lansing. The concentration of mercury in the sediments, presumably from urban and industrial drift from the metropolitan area located to the southwest of the lake, has been found to be in the range of 0.5-1.0 ppm by analyses in the Water Chemistry Laboratory of the Institute of Water Research at MSU. We propose to monitor the mobilization of these elements to water of the lake and to the downstream environment before, during, and after dredging because of their potential toxicity.

Scrutiny of the biological food chain to fish is an important aspect of this work for the reason that the sport fishery of Lake Lansing is a substantial recreational resource. Presently existing detrimental effects on the fishery due to excessive primary production of macrophytes is suggested by the work of Roelofs (1958). Since condition of fish populations in the lake may have changed substantially in the years since that report, a pre-dredging assay of them will be done in the fall of 1978. Since qualitative and quantitative data on the zooplankton and benthic invertebrates of the lake are entirely lacking, pre-dredging analyses on these associations will be done as well. Similar post-dredging studies will be done for comparison.

METHODS OF LIMNOLOGICAL SAMPLING

THE LITTORAL ZONE COMMUNITIES

The state of the aquatic macrophyte community of the lake will be described by 1) a map of the distribution of the dominant species in the basin at the height of the growing season, 2) detailed physiognomic profiles of the plant communities along six transects through the littoral zone, and 3) an estimate of the annual maximum standing crop of this vegetation in the lake. Each of these aspects of the study will include the dominant filamentous algae of the lake as a component of the macrophytic vegetation. The work will be done in August and early September of 1978, and again in years following completion of the dredging.

The map of distribution of dominant species will be made from the observations of teams of swimmers working over contour intervals with compass reference to landmarks. The 0-1 m, 1-2 m, 2-3 m, and 3-4.5 m depth intervals will be mapped in succession. Observations on the lake in recent years indicate that particular vegetational associations exist to a depth of 4.5 m in the basin. In particular, the extent of stands of emergent vegetation and submersed characean meadows, *Elodea-Najas* communities, and mixed communities of potamogetons, milfoils, and *Vallisneria* will be documented.

Figure 10 shows the locations of transects that will be made through the littoral communities from shore to the 4.5 m contour. These locations have been selected over the major sediment types in the lake, and through areas in which dredging will be done. Reference to the figure will show that two of the transects (1 and 6) will be done over peat, two will be done over marl (2 and 3), and one will be done over alternating peat and marl sediments (5), and the remaining transect (4) will course over sand. Profiles of the vegetation showing the location of species and the height of component species will be made at each location.

For the purpose of sampling to estimate the maximum macrophyte biomass during the pre-dredging year of 1978, the lake has been sectured as shown in Table 1.

TABLE 1. THE ALLOCATION OF NUMBERS OF SAMPLES FOR PRE-DREDGING BIOMASS ESTIMATES OF LITTORAL ZONE VEGETATION IN LAKE LANSING.

Stratum	Area (ha)	of Samples	on Transects
		SOUTH BASIN	
0-1 m	4.9	3	1
1-2 m	6.1	4	1
2-3 m	4.9	3	1
3-4.5 m	2.8	2	1
		NORTH BASIN	
0-1 m	23.5	17	5
1-2 m	40.9	26	5
2-3 m	57.9	37	5
3-4.5 m	14.6	8	5
Totals	155.6	100	

For the lake as a whole, the areas of 0-1 m, 1-2 m, 2-3 m, and 3-4.5 m contour intervals are in the approximate ratio of 2:3:4:1. The total area of the littoral zone of the south basin is approximately 1/7 of that area in the north basin. The total number of samples to be taken has been proportioned within basins and within contour intervals to reflect these relationships. For each sample, an area of vegetation having a maximum biomass will be selected. A cylinder 60 cm deep with a cross-sectional area of 0.1 m^2 will be placed over the vegetation and pushed into the sediments. The plant material

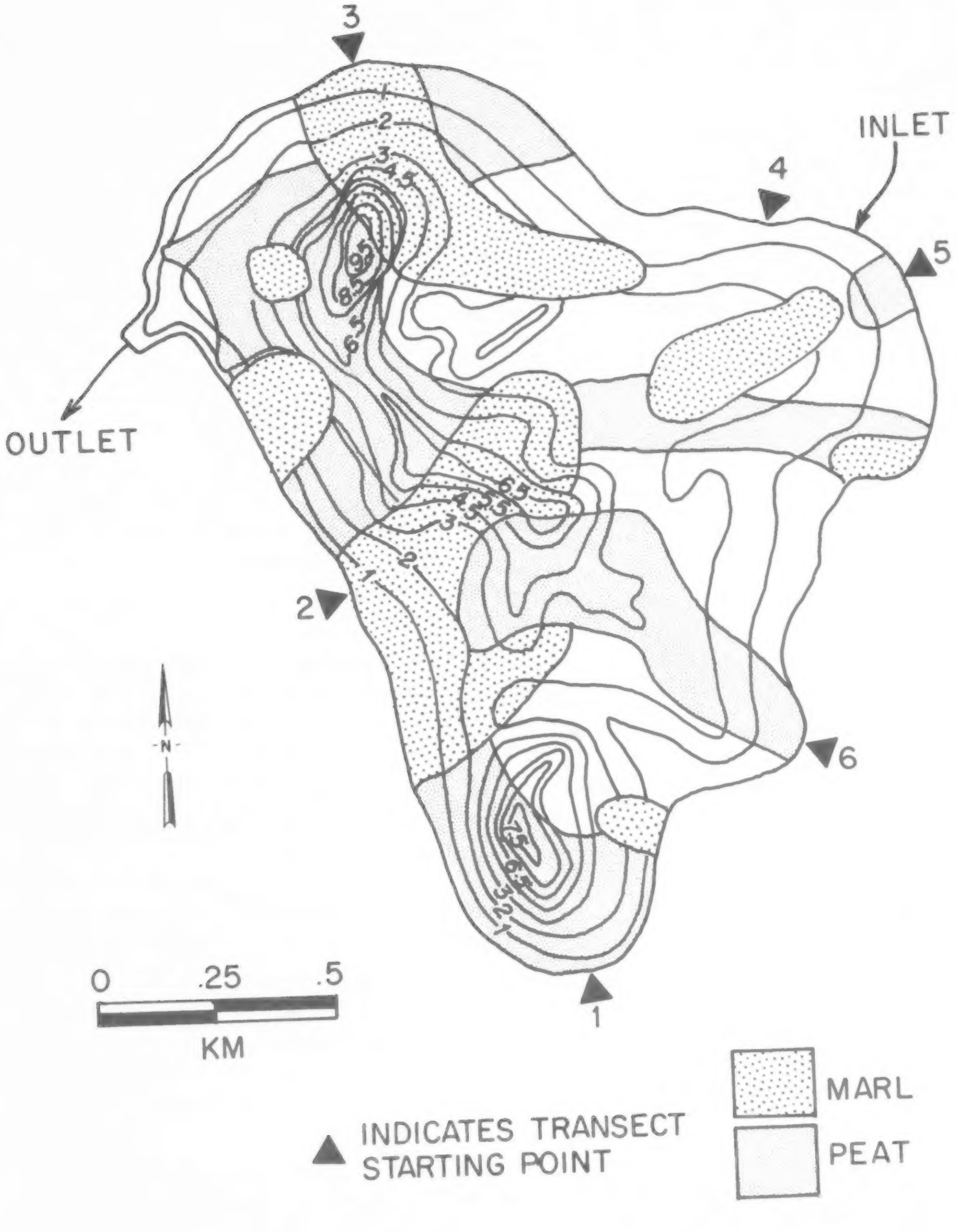

3
INLET
4
5
OUTLET
2
6
1
N
0
.25
.5
KM
MARL
PEAT
INDICATES TRANSECT STARTING POINT

Figure 10. Distribution of marl and peat in Lake Lansing sediments (after Snell, 1970) and starting points for sampling transects that will traverse the littoral to 4.5 m.

will be removed by hand. It can be noted in Table 1 that each contour interval of the six littoral transects will be sampled in this way to incorporate the feature of biomass into the profiles of the vegetation obtained along these lines.

The map of the distribution of the vegetation cited above will be used to select sampling areas outside of the transects. An estimate of percent cover within each contour interval will also be available from that map. After the samples of vegetation have been individually washed free of sediments, dried at 105°C and ashed at 530°C, an estimate of biomass (biased toward the maximum) will be obtained for each contour interval, and the littoral zone as a whole, from the relationships of mean sample weight and percent cover. As the plant material is handled to obtain weights, the filamentous algae will be separated out and treated as a distinct component.

A new bathymetric map will be made in the post-dredging years. The new distribution of littoral vegetation will be superimposed upon it by using the techniques for mapping described above. Transects 1 through 6 will be redescribed. The size of the annual maximum standing crop of aquatic macrophytes and filamentous algae will be estimated from the same total number of samples using the same approach as applied in the pre-dredging year.

THE PLANKTON AND THE AQUEOUS MEDIUM

The littoral and pelagial portions of Lake Lansing will be sampled separately to obtain information relevant to the effects of dredging on the system. A pelagial station will be established over each of the two deep holes on the basin shown in Figure 2. The 0.5 m, 1.5 m, 2.5 m and 3.75 m contours in Transects 1 through 6 of Figure 10 will serve as littoral stations in the pre-treatment year of 1976. These stations will be located by appropriate triangulation and will serve as sampling points in the dredging and post-dredging period as well.

Three 2 L water samples will be taken from mid-depth at each station on a littoral transect. These will be composited by contour, resulting in four samples of 36 L each per sampling date. At the pelagial stations, composite samples representing the epilimnetic and hypolimnetic regions will be obtained. Three 3 L samples from each of 0.5 m, 1.5 m, 2.5 m and 3.75 m depths will compromise the former. Three samples at each depth beginning at 5.5 m and spaced 1 m apart to within 1 m of the bottom will be taken as representative of the latter. Aliquots of these composites will be consigned to various analyses.

The phytoplankton of a sample will be described by making a list of the dominant species, counting the cells per unit volume of water for those species, converting all counts to cell volume by species, and measuring the chlorophyll a per unit volume. The kinds and numbers of zooplankters will similarly be recorded. Aliquots will be used for chemical analyses essential to the nutrient budget of the lake as described below.

Vertical temperature and oxygen profiles will be obtained at these same stations using a YSI model 54 unit standardized for dissolved oxygen against the azide modification of the Winkler method. In the open-water period of the year, these profiles will be obtained at dawn-dusk-dawn on successive days so that diurnal and nocturnal excursions in oxygen can be recorded. During the period of ice-cover on the lake, these measurements will be made near mid-day principally to follow the time-course of oxygen depletion in the tropholytic zone. Simultaneous treatment of pH and alkalinity will be used (along with temperature) to develop carbon dioxide relationships in the profiles.

A surface and submarine cell of a Schueler photometric system will be used to obtain percentages of surface light transmission that can be transformed into vertical extinction coefficients for the lake.

THE NUTRIENT BUDGET

While phosphorus and nitrogen are the nutrients primarily implicated in controlling the state of primary production in lakes of our region, additional elements are of interest in Lake Lansing for reasons expressed earlier in this paper. These are Si, Ca, Mg, Na, K, As, Cu, Hg, total organic carbon (TOC), and dissolved organic carbon (DOC). These and nitrogen and phosphorus will be measured as inputs to the lake, within the lake, and as outputs to the downstream environment. The analytical methods will be those recommended by EPA-Water Programs under Title 40, Part 136, Federal Register, Volume 41, No. 232, pages 52780-52786 on December 1, 1976.

Figure 1 of this document shows the watershed of Lake Lansing to be relatively small (8.42 km^2) and poorly drained in terms of surface point discharges to the lake. Wetlands in the watershed trap a significant portion of the precipitation as indicated by the arrows on that figure. Adjustments between the elevation of the surface of the lake and the water standing in the wetlands are made largely by seepage through glacial till, with intermittent overland flow. Young et al. (1974), using data that was gathered early in the 1970's estimated the nominal retention time of the lake to be 1.6 years.

In the fall of 1976, a previously existing concrete overflow structure on the outlet of the lake was replaced in accord with a newly established legal lake level. The summer lake level is now regulated at 2.54 cm above the previous standard; the drawdown winter lake level is set at 12.7 cm below the previous standard. The past year has been unusually dry for this region. There has been no surface discharge from the lake since July, 1976; the surface of the lake has been 15-20 cm below the established outfall elevation. Estimates of the rate of water removal during dredging suggest that the lake level will be depressed by that process. Sampling to obtain the best possible estimates for a nutrient budget will be done with due regard to the hydrologic condition of this system.

Changes that occur in the lake in the amounts of the elements of concern will be determined from weekly samples in the interval from mid-March through October, and by two-week sampling over the remainder of the year. A composite sample from 18 locations over each of the 0.5 m, 1.5 m, 2.5 m and 3.75 m contours will be obtained each date for the littoral zone. Over the two depressions in the basin, a composite of three samples each will be made from the above depths and at 5.5 m, 6.5 m , 7.5 m, 8.5 and 9.5 m. Following laboratory analysis, concentrations in these samples multiplied by the volumes of water within the sampled contour intervals will yield estimates of the quantities present in the basin. Volumes of water within contours will be obtained from depth-volume curves like that of Figure 5 of this document adjusted by mapping for changes due to dredging. Because of limited overland flow to or from the basin, in-lake sampling will be an important aspect of the budget and is likely to show significant changes due to dredging, if they occur.

Precipitation could account for a significant fraction of the phosphorus and nitrogen budgets of the lake. Annual precipitation is on the order of 60-70 cm (Young *et al.*, 1974). On the basis of the work of Chapin and Uttormark (1973), 0.01-0.1 g P m^{-2} yr^{-1} and near 1 g combined inorganic N m^{-2} yr^{-1} would be expected to fall on Lake Lansing. Vollenweider (1968) considered values close to these to be dangerous from the standpoint of eutrophicational control in shallow basins like Lake Lansing (<5 m mean depth).

Precipitation on the surface of the lake will be measured with a Weather Measurement Corporation P511-E heated gage that is acceptable for both rain and snow. The gage will be coupled with a P522 clock-drive long-term event recorder to obtain a record of the occurrence and quantity of precipitation. Evaporation will be measured with a standard WMC E810 manual evaporation station. Precipitation for nutrient analyses will be collected by a custom built 2 m diameter polyethylene funnel set on an insulated plywood box that can be heated to 5¼-10¼C in winter. The funnel will lead to a 40 liter collection carbuoy. The instruments will be mounted in a protected area on the shore of the lake at the Lake Lansing Yacht Club. They will be attended each day a crew is in the field, except that precipitation for nutrient analyses will be brought into the laboratory within 8 hours of each event. Incident solar radiation for the study will be determined by planimetry of curves from a recording Epply pyrheliometer maintained at MSU's South Farm climatological station located approximately 5 km from the lake.

Six culverts join Lake Lansing with adjacent wetlands. Movement of water through these tends to coincide with periods of heavy rainfall and rapid snow-melt. Weekly measurements will be made of the volume of flow at these points using hydraulic cross-sections in the culverts and estimating current velocity with a Pygmy-type water current meter. Samples for analytical analyses will be taken when flow occurs. If periods of unusually high runoff occur during the study, these inputs will be sampled 2-3 times weekly. The intervals of the general schedule will also be shortened if removal of water from the lake by dredging is observed to significantly alter the rates of flow from surrounding wetlands. Return water from the dredged materials disposal sites will be sampled when it discharges, and results will be included in the budgets for the lake.

The occurrence of an outfall from the lake will be checked on each day a working crew is in the field; for the five-year program, the longest interval between observations would be one week. If an outfall occurs, it is likely on past history to be intermittent. When a discharge is first noted, its rate (e.g. m^3/sec) will be determined. The volume of water in the lake above discharge elevation of the spillway will be estimated from depth at the spillway and surface area of the lake at that elevation. The time required for the lake to return to base level will be computed without regard for losses by evaporation. A sampling schedule will then be constructed such that t_0 is the time of first observation of discharge (and sampling), t_0-t_1, t_1-t_2, t_2-t_3, t_3-t_4 and t_4-t_5 are equal intervals, and t_5 is the time when base level should be reached. At each time of measurement and sampling collection after t_0, the time required to reach the base level will be recalculated. If the result is $< t_5$, the sampling will progress as planned; if the result is $> t_5$, a new sampling schedule like the first will be constructed. Under no circumstances will the interval between samples be longer than 7 days.

NUTRIENTS IN THE SEDIMENTS

The exposure of sediments of different solid-phase constituents and interstitial water equilibria will occur as dredging is done in Lake Lansing. Thus nutrient availability in the absorptive layer for rooted aquatic plants, or quantitative aspects of concentration gradients in the region of the water-sediment interface could be changed by dredging. These aspects of the nutrient regime will be studied by use of the sample collecting device of Mayer (1976) shown in Figure 11. It depends in principle upon diffusive equilibration of solutes between interstitial water and the content of removable dialysis bags. These samplers will be used to measure concentrations of nutrients in 10 cm strata from 50 cm above the sediments to 50 cm below their surface.

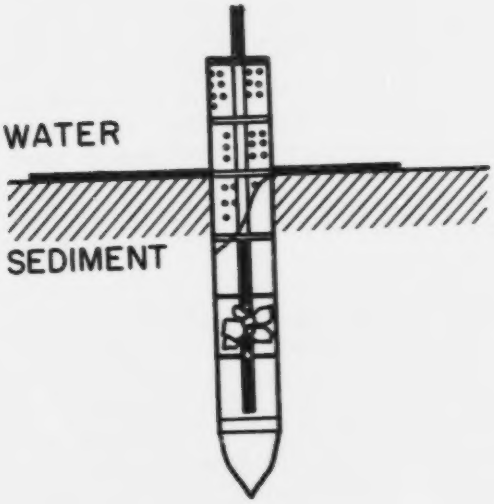


Figure 11. Diagrammatric presentation of interstitial water sampler with one dialysis bag in place (from Mayer 1976).

Equilibration times will be determined in the laboratory prior to sampling with Lake Lansing sediments at sampling temperature according to the methods of Mayer (ibid). A 50 cm depth in the sediments has been chosen on the basis of our observations of root penetration by macrophytes in the lake, and on the strength of the work of Hynes and Grieb (1970) who showed movement of phosphorus to overlying water from undisturbed anoxic sediments occurred from a depth of at least 10 cm in 2 to 3 months.

Mayer sampling will be done in the mid-June and late-July interval. The purposes of this timing are to obtain data from year to year under relatively constant conditions of temperature, and after thermal stratification has set-up in deep portions of the basin. The locations of sampling will be in those areas that are selected in the final plan for dredging. Sediments to be dredged have been designated by Snell (1970; Figure 10 here) as marl, peat (gyttja), and sand. Five samplers will be set at each of three permanently located stations within each sediment type in each year. Individual samplers will be set in macrophyte-free portions of the littoral. After equilibration, the samples from each of ten depths will be composited for analyses by station. Soluble phosphates (unreactive P, reactive P, total p), inorganic nitrogen (NO_2-N, NO_3-N and NH_3-N) and silica (SiO_2) will be measured by techniques used in this study for water analyses.

BENTHOS AND FISH POPULATIONS

A Ponar dredge will be used to obtain samples of the benthic communities of Lake Lansing. Eight sampling stations will be used per date of sampling; one over each of the two deep holes in the lake and one on each of the six transects shown in Figure 10. Samples from a transect will be taken at 1978 depth 2.5 m. These positions will be permanently located by triangulation using landmarks. In the post-dredging period, these stations as well as stations on the newly located 2.5 m contour will be sampled. Three samples will be taken at each sampling point. Periphytic invertebrates and those on the sediments will be included in samples from the littoral region. Those organisms taken by the dredge and retained on a U.S. Standard No. 30 sieve, will constitute a sample. They will be enumerated and weighed by type so that estimates of biomass will result.

Electrofishing gear of the Department of Fisheries and Wildlife at MSU will be the principle tool for sampling fish populations. Seines, gill nets and traps will be employed as supplemental devices. Species of fishes existing in the lake will be identified. Their relative abundance in collections will be determined. The individuals will be sexed and scale samples will be utilized for determination of age and rate of growth. Coefficients of condition will be calculated from measurements of length and weight. Estimates of the size of populations will be made from records of catch per unit effort, and from tag and recapture studies.

ECOLOGICAL IMPACTS ON DREDGED SPOILS DISPOSAL SITES

Hydrous materials from the bottom of Lake Lansing will be lifted by a hydraulic dredge to diked disposal areas in the vicinity of the lake. In the early stages of planning the project, close-by marshes in private ownership were considered as the most probable sites for disposal. Cole and Prince (1976), responding to the need for an ecological evaluation of the marshes, classified most of the wetland areas in the immediate vicinity of the lake. One hundred twenty-four ha of wetlands were categorized according to the scheme of Golet and Larson (1974). Over time however, the marshes have been excluded from use as disposal sites. Dredgings will be lifted to uplands on the watershed. At the time of this writing, there is some question as to the specific locations that will be used. Whatever the final choice, vegetational changes and changes in use by animal species will be recorded for the sites over the five or more years of the study. The discharge water from disposal sites will very likely course through marshes on its return to the lake. Where this occurs, the impact on the marshes will be measured in terms of retention of nutrients and sediments, and the effects of these on the vegetation and animal and human use.

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DILUTION EFFECTS IN MOSES LAKE

by

E. B. Welch and C. R. Patmont*

INTRODUCTION

The effects of adding low nutrient dilution water to eutrophic lakes, for the purpose of reducing their algal content, are twofold and lead directly from the dynamics of continuous cultures. By reducing the inflow nutrient concentration, the maximum biomass possible in the reactor vessel of a continuous culture is likewise reduced, and, by increasing the water exchange rate, nutrients and algal biomass are more rapidly washed out of the reactor, preventing an accumulation. Since the concentrations of nutrients and biomass in the reactor are the critical parameters in lakes, the controlling factors in continuous cultures are often analogous in lakes.

The effect of inflow concentration follows from Vollenweider's (1969) model for the steady state phosphorus concentration;

$$\bar{P} = \frac{L}{\bar{Z}(\rho+\sigma)}$$

where L is the areal loading, $\bar{Z}$ is the mean depth, and ρ and σ are the coefficients for the flushing (water exchange) and sedimentation rates, respectively. Clearly if the flushing rate (ρ) can be increased proportionately more than the areal loading (L), which is the result of adding water with low nutrient concentration, then the steady state P concentration ($\bar{P}$) should decrease. That should theoretically reduce the potential biomass of algae in the lake. If enough water can be added so that the exchange rate approaches the growth rate of the algae, then biomass reduction can occur through washout of cells at a rate that exceeds the growth rate.

Both mechanisms were thought to have potential in Moses Lake, but mainly that of a reduction in inflow nutrient concentration (Welch, et al., 1972). Because the bluegreen algal component (mostly Aphanizomenon) was found to grow at a rate of at least 0.5 day^{-1}, water exchange rates approaching that magnitude were thought to be necessary to control biomass. However, an additional observation in the earlier bioassays was that the growth of bluegreen algae was poorer and that of diatoms better as the percent of dilution water increased. It was of interest, from the standpoint of food chain energy transfer and nuisance conditions, to see if such a species change occurred on a large scale with the influx of Columbia River dilution water.

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This paper describes the results of adding Columbia River water to an arm of Moses Lake in Eastern Washington during the spring-summer of 1977. Because of intensive monitoring of trophic state indicators during 1969-70, the relative improvement in water quality in 1977 could be evaluated with respect to a normal year. The characteristics of Moses Lake are shown in Table 1.

Water was diverted through Parker Horn of Moses Lake from a large irrigation system via the Eastlow Canal and Rocky Coulee Wasteway (Figure 1). The proposal was to add water at about 32 m^3 sec^{-1} for 10 days in the spring. Based on studies with a physical hydraulic model (Nece, *et al.*, 1976), the results of which compared closely with those from a simple continuity model, that magnitude of water input was considered adequate to maximize the reduction of total phosphorus in Parker Horn. Depending upon the rate of return to pre-dilution lake P, a second and possibly a third 10-day application was proposed during the summer.

That the concentrations of nutrients in Columbia River water relative to the lake and highly favorable for such a dilution project is shown in Table 2. Of interest is the much higher concentration of TP in the lake, versus Crab Creek, the natural inflow. The high lake values are a result of wind blown algae into upper Parker Horn, resuspension of sediment by wind, and excretion by carp.

While water was not available in the exact quantities or at times proposed, nevertheless, climatic conditions and continual coordination with U.S. Bureau of Reclamation personnel at Ephrata, Washington provided an experimental design that was adequate for the purpose intended, which was to;

1) determine if TP in Moses Lake could be controlled with dilution water as predicted by physical and mathematical models,

2) determine if the algal blooms (chl *a*) could be reduced in proportion to the reduction of TP and if water clarity could be improved,

3) observe if a species shift from bluegreen algae to diatoms as a result of the TP reduction occurred, and

4) estimate what the optimum pattern of dilution water input would be to control algal crop to about 20 $\mu g\ l^{-1}$ chl *a* and TP to 50 $\mu g\ l^{-1}$.

PROCEDURE

ANALYSIS

Sampling of water for the determination of total phosphorus (TP), Ortho P (OP), nitrate nitrogen (NO_3-N), total N (TN), chlorophyll *a* (chl *a*) and plankton counts was conducted by pumping water from a depth of about 0.4 m at 7 horizontal transect sites (Figure 1). At a midpoint of each transect, profiles of DO, pH, temperature and specific conductance were also determined, as

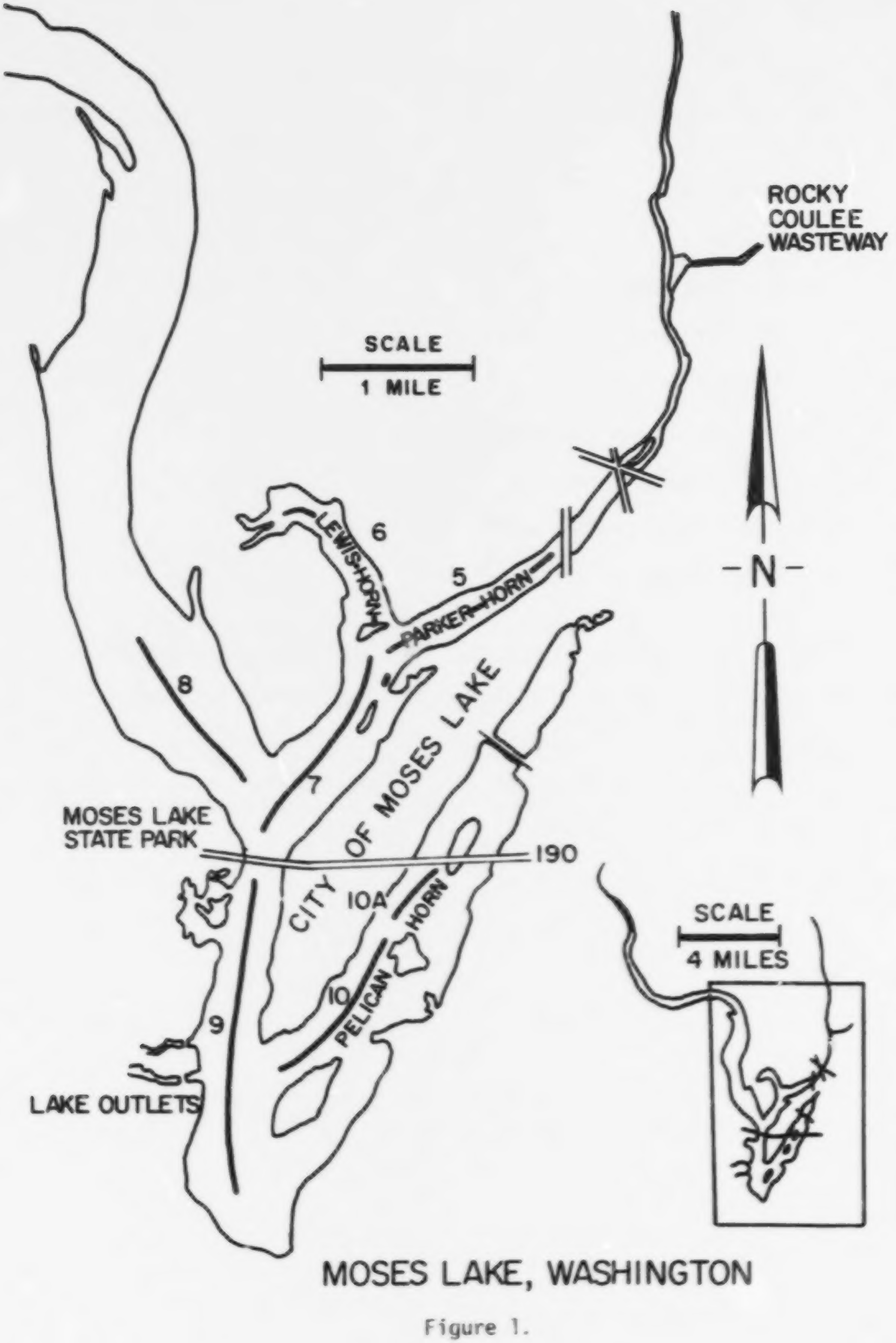
TABLE 1. LIMNOLOGICAL CHARACTERISTICS OF MOSES LAKE (1969-70).

Area	2,753 hectares
Mean Depth	5.6 meters
Volume	153.7×10^6 m^3
Flushing Rate	1.8 yr^{-1}
Nitrogen Load	18.7 g m^{-2} yr
Phosphorus Load	2.1 g m^{-2} yr^{-1}
Average Inflow P Concentration	190 µg l^{-1}
Chl a (summer mean)	100 µg l^{-1}
Secchi Disk (summer mean)	0.5 m
Total P (summer mean)	135 µg l^{-1}
Ortho P (summer mean)	50 µg l^{-1}

TABLE 2. CONCENTRATIONS OF NUTRIENTS AND Chl a IN MOSES LAKE AND INFLOW WATER DURING 1969-70 IN µg l^{-1}.

	Parker Horn	Columbia River	Crab Creek
Total P	178	30	92
Ortho P	35	17	38
Nitrate N	74	3	843
Ammonia N	--	--	--
Total N	(1480)	80	(1180)
Chl a	148	(2)	16

( ) estimated because data lacking


ROCKY
COULEE
WASTEWAY
SCALE
1 MILE
LEWIS HORN
6
5
PARKER HORN
N
8
CITY OF MOSES LAKE
7
MOSES LAKE
STATE PARK
190
10A
HORN
PELICAN
10
9
SCALE
4 MILES
LAKE OUTLETS

MOSES LAKE, WASHINGTON

Figure 1.

well as Secchi disk depth. Samples were also collected in the profile for the determination of nutrients and chl a, but were not used in this analysis. The horizontal samples were used to compare with values from 1969-70, determined from similar collections.

Samples were collected weekly, except during the three dilution periods when they were collected every other day. Filtration and other sample preparations were conducted at facilities at Moses Lake, but nutrient and chl a analyses were performed the following day at Seattle. Procedures for nutrient analysis were largely those from Strickland and Parsons (1968) with the following characteristics;

1) phosphate - ammonium molybdate heteropoly blue complex

2) total phosphorus - persulfate digestion

3) nitrate - copper cadmium reduction column

4) organic nitrogen - u.v. light oxidation

5) chlorophyll a - fluorometric analysis of acetone extracts

6) DO, pH, temperature and specific conductance - Martek water quality analyzer

THE 1977 ADDITION

Dilution water was added to Parker Horn during three periods in 1977. The periods, the average inflow rates for Columbia River water plus Crab Creek, Crab Creek alone, and the resulting water exchange rates, are given in Table 3. The first period lasted about 1.5 months at a rate of 0.25 day^{-1}. The second period was only two weeks, and the last was nearly a month.

TABLE 3. AVERAGE FLOWS OF CRAB CREEK AND DILUTION WATER (COL. RIVER) AND AVERAGE WATER EXCHANGE RATES FOR PARKER HORN DURING THREE PERIODS IN 1977.

	m^3 sec^{-1}		
	Total Inflow	Crab Creek Base Flow	Water Exchange Rate, Day^{-1}
3/20-5/7	34.0	0.40	0.25
5/22-6/4	11.8	1.34	0.09
8/14-9/10	19.8	2.50	0.15

These rates of exchange were calculated for Parker Horn alone, which includes stations 5 and 7. They will be used to compare the observed changes in P, N, chl a and conductance with what would be expected from the following equation:

$$C_t = Ci + (Co - Ci)\ e^{-kt}$$

where C_t is the concentration at time t, Ci is the inflow concentration, Co is the initial lake concentration, and k is the water exchange rate. the expected levels in Parker Horn were calculated on a weekly basis.

RESULTS

EFFECTS OF DILUTION

The effect of the three treatments of dilution-water addition was to reduce conductance, P, N, and chl a content and increase the Secchi disk depth. The average values from May through August, 1977 are compared to similarly calculated values during 1960 and 1970 in Figures 2 through 5. An interesting point is that water quality improvement was also observed well into Lewis Horn, the main lake and into lower Pelican Horn. Thus, the effect of adding dilution water to Parker Horn via Rocky Coulee Wasteway is much more extensive than just in upper Parker Horn. However, the reduction in total P and chl a was greatest in upper Parker Horn, as would be expected, since that volume would have been most completely replaced by Columbia River water. The actual Secchi disk depth, on the other hand, was greatest at the lower lake stations, although relative improvement was still better in Upper Parker Horn.

The overall better visibility (greater Secchi disk depths) in the lower lake is related to the actual lower chl a values (Figures 4 and 5), even though percent improvement was not as great as in the upper lake. There are two processes happening that could account for this trend of downlake decreasing chl a and increasing clarity. First, the degree of wind-induced turbulence would decrease vertically in downlake progression. With less turbulence, the sedimentation of algae and other particulate matter from the water column would increase. Second, the south winds tend to blow floating blue green algae up into upper Parker Horn. That this is the factor operating is indicated by the similar P content in Parker Horn and the lower lake, while at the same time there is the trend of changing chl a and Secchi disk values, which are clearly inversely correlated.

The important observation from the management standpoint is that dilution water entering upper Parker Horn also markedly improved the lower lake. The area of lake beneficially affected was greater than expected. This can be shown numerically in Table 4 where the percent improvement in P, chl a, conductance and Secchi disk depth with respect to expected goals was nearly as great (or greater) in the lower lake (Station 9) as in Parker Horn (Station 7). Of course, as indicated in Figures 3, 4 and 5 the actual percent reduction in P and chl a and percent increase in water clarity in upper Parker Horn (Station 5) was more than that in the lower lake largely due to the more complete replacement of Parker Horn water with Columbia River water, as noted earlier.

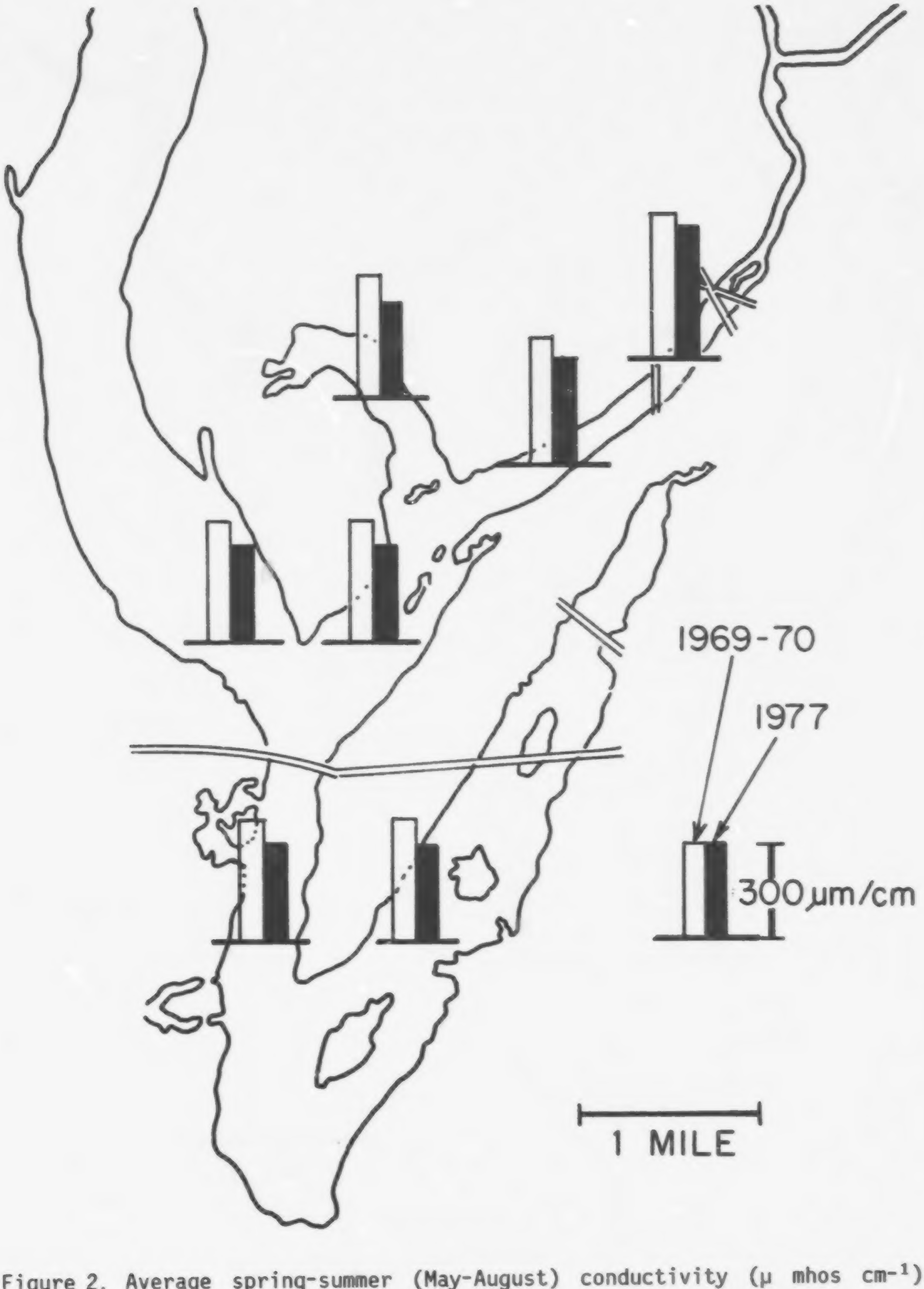

1969-70
1977
300 μm/cm
1 MILE

Figure 2. Average spring-summer (May-August) conductivity (μ mhos cm^{-1}) in Moses Lake in 1969-70 and in 1977.

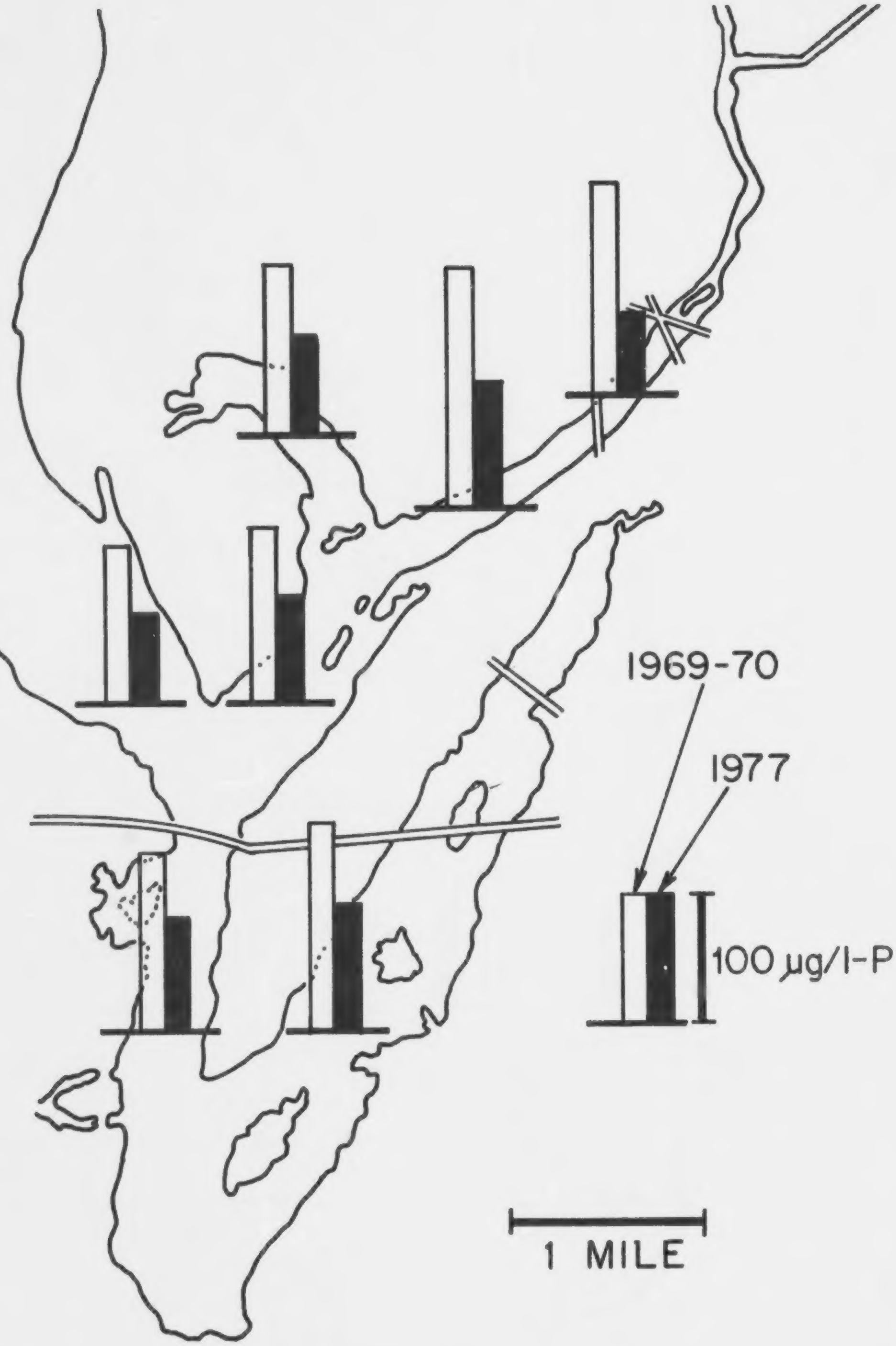

1969-70
1977
100 µg/l-P
1 MILE

Figure 3. Average spring-summer (May-August) total phosphorus content (µg l^{-1}) in Moses Lake in 1969-70 and in 1977.

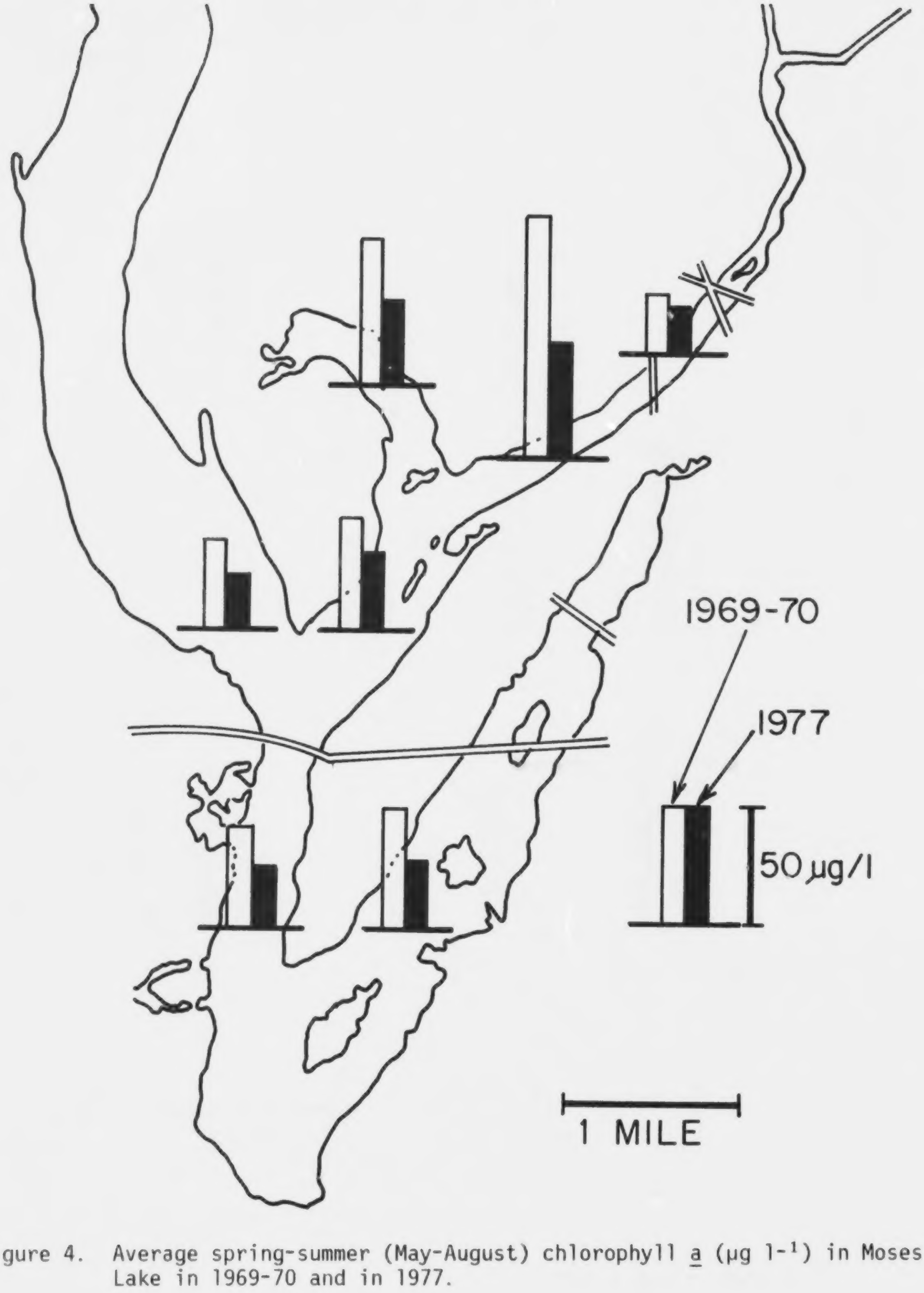

1969-70
1977
50 µg/l
1 MILE

Figure 4. Average spring-summer (May-August) chlorophyll a ($\mu g\ l^{-1}$) in Moses Lake in 1969-70 and in 1977.

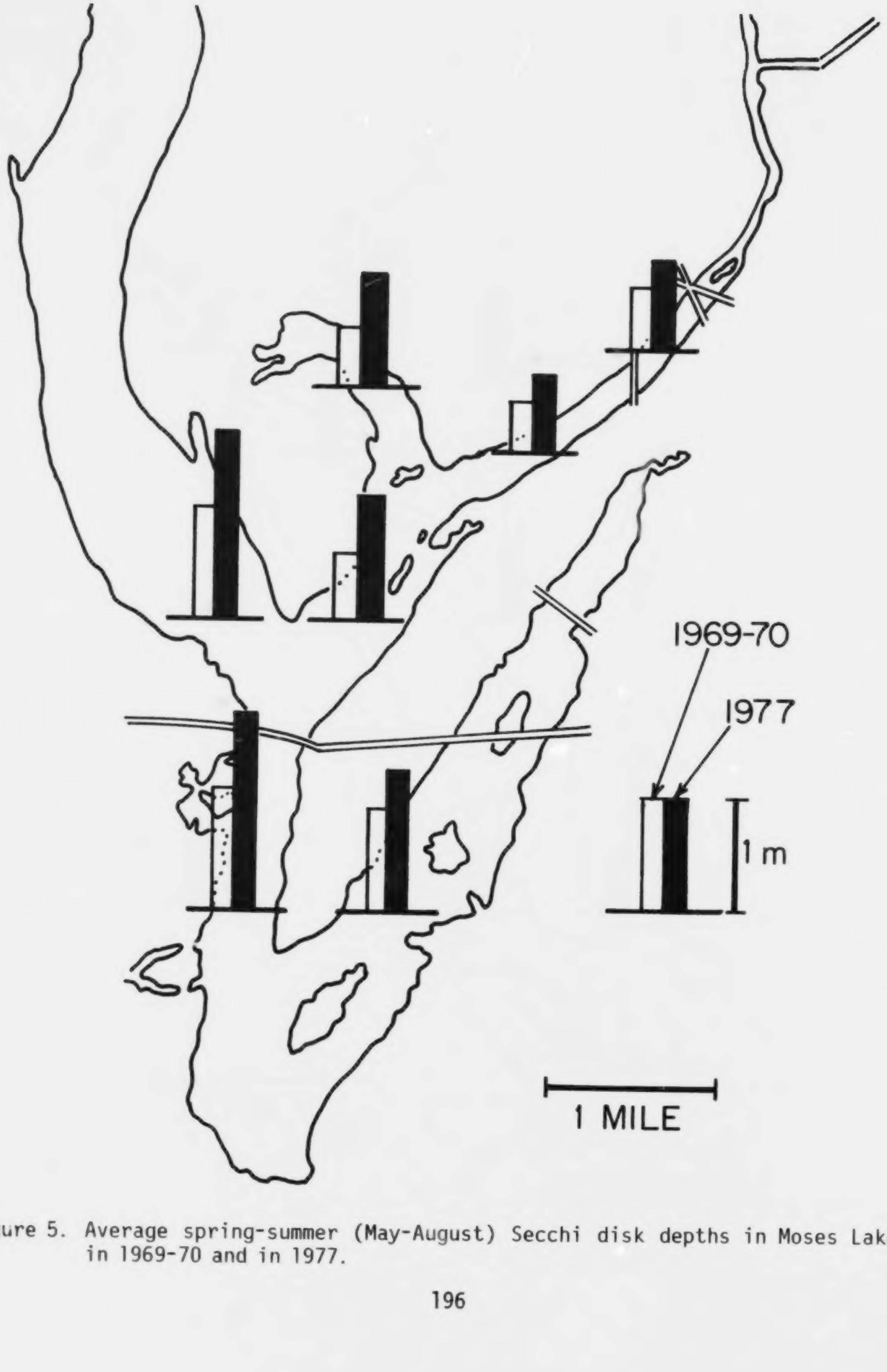

1969-70
1977
1 m
1 MILE

Figure 5. Average spring-summer (May-August) Secchi disk depths in Moses Lake in 1969-70 and in 1977.

The interesting and apparently anomalous result of the dilution demonstration is further indicated in Table 4. Initially it was proposed that in order to reduce algal content (chl a) and improve clarity (Secchi disk depth), P content must first be reduced. It has been hoped that P would be lowered to an average of 50 μg l^{-1} or less which should have resulted in a chl a content of 20 μg l^{-1} or less, based on the predictive equation of Dillon and Rigler (1974). According to the average levels of P (78 and 91 μg l^{-1}) the expected mean chl a content should have been about 40 and 50 μg l^{-1} respectively. While it can be argued that the system was changing too much from the three separate inputs of water for a valid use of the Dillon and Rigler equation, which is more representative of an equilibrium condition, the results indicate that chl a was reduced far more than could be expected from P reduction alone and came much closer to attaining a goal in spite of the poorer improvement in P. Likewise Secchi disk depth was increased far more than would have been expected based solely on chl a, and even exceeded the goal in the lower lake.

TABLE 4. AVERAGE TOTAL P, TOTAL N, Chl a AND SECCHI DISK DEPTH IN 1977 (MAY-AUGUST) COMPARED TO 1969-70 FOR PARKER HORN (STATION 7) AND THE LOWER LAKE (STATION 9).

	Total P	Conductance	Chl a	SD (m)
PARKER HORN (STATION 7)				
1977	78	311	29	1.2
1969-70	135	403	73	0.6
Goal	50	300	20	1.5
% Improved*	67	89	83	67
LOWER LAKE (STATION 9)				
1977	91	309	24	1.9
1969-70	135	402	44	1.0
Goal	50	300	20	1.5
% Improved*	52	91	83	180

* Concentrations are in μg l^{-1} and conductance in μmhos/cm. Percent improvement for Total P, Conductance, and Chl a was calculated by the difference between 1960-70 values and the goal for restoration. Thus, percent improved refers to the extent to which the goals were attained in 1977.

The changes in the variables measured throughout the spring and summer of 1977 are shown in Figure 6 through 9. The values reported here are from the horizontal collections at a depth of about 0.4 m. While it is apparent that the P levels in Moses Lake north and south of the I-90 bridge were lower

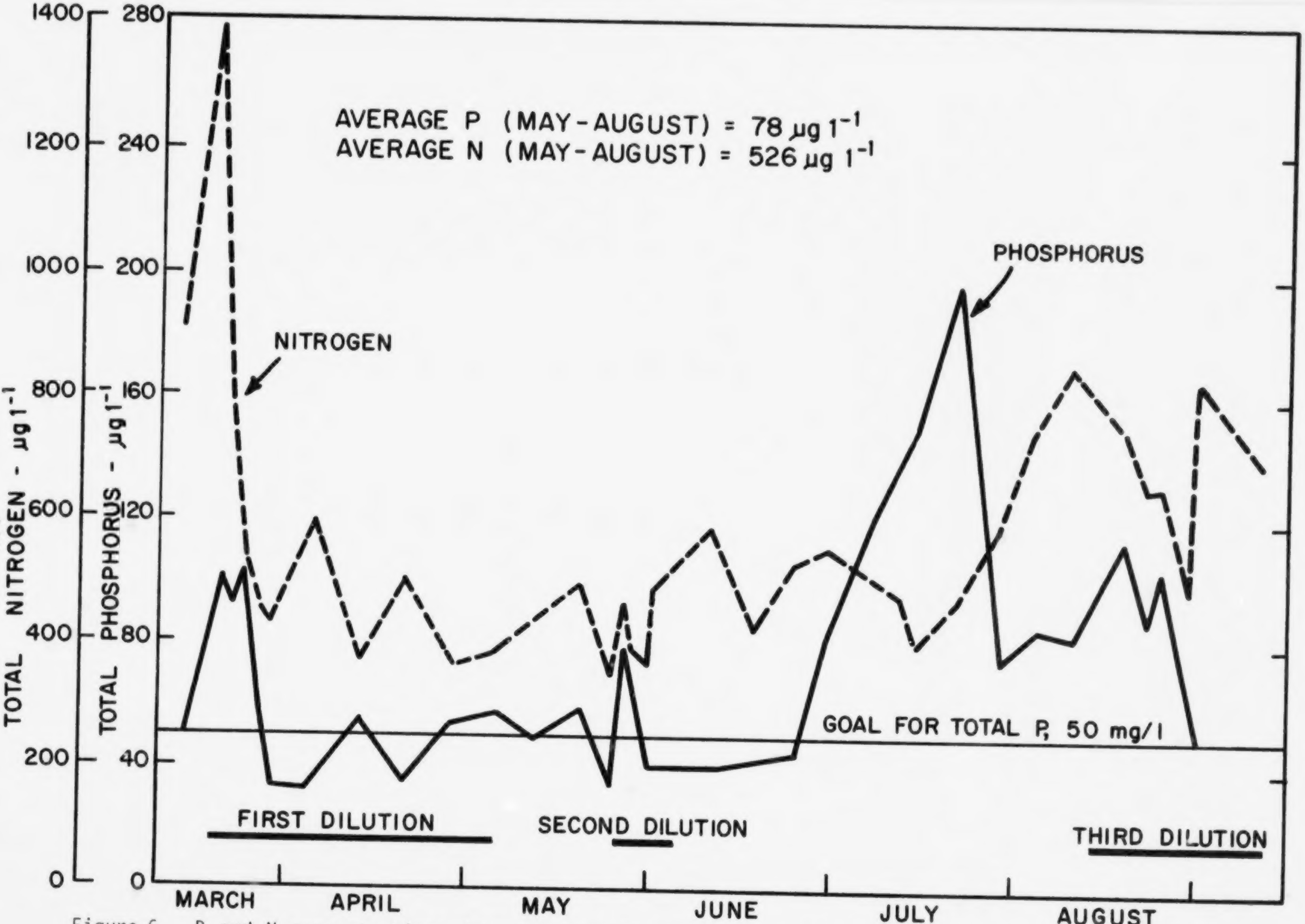


Figure 6. P and N concentrations from horizontal transects at Station 7, Moses Lake, North of the I-90 bridge, 1977.

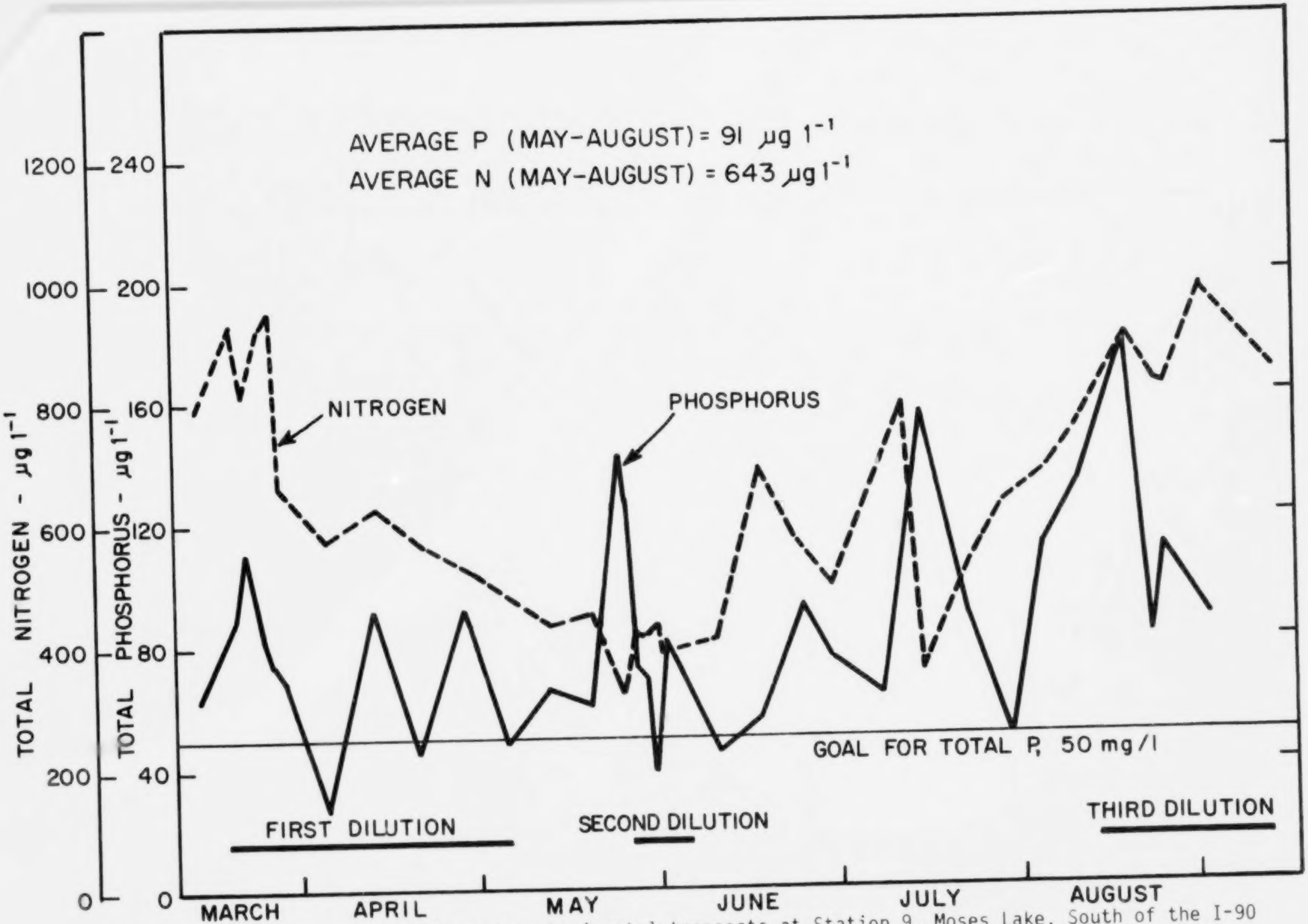


Figure 7. P and N concentrations from horizontal transects at Station 9, Moses Lake, South of the I-90 bridge, 1977.

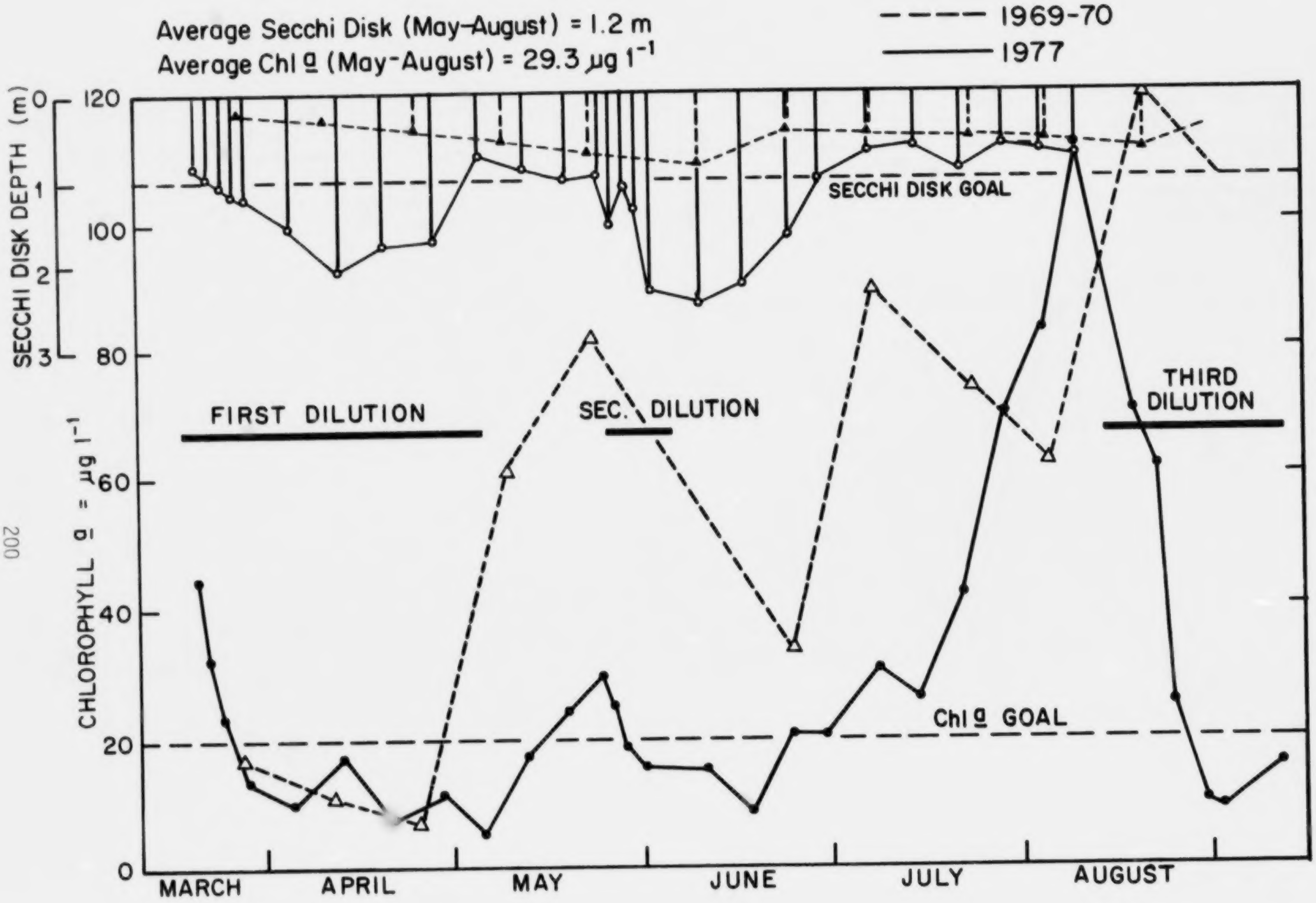


Figure 8. Chlorophyll a from horizontal transects and Secchi disk measurements at Station 7, Moses Lake, North of the I-90 bridge during 1969-70 and 1977.

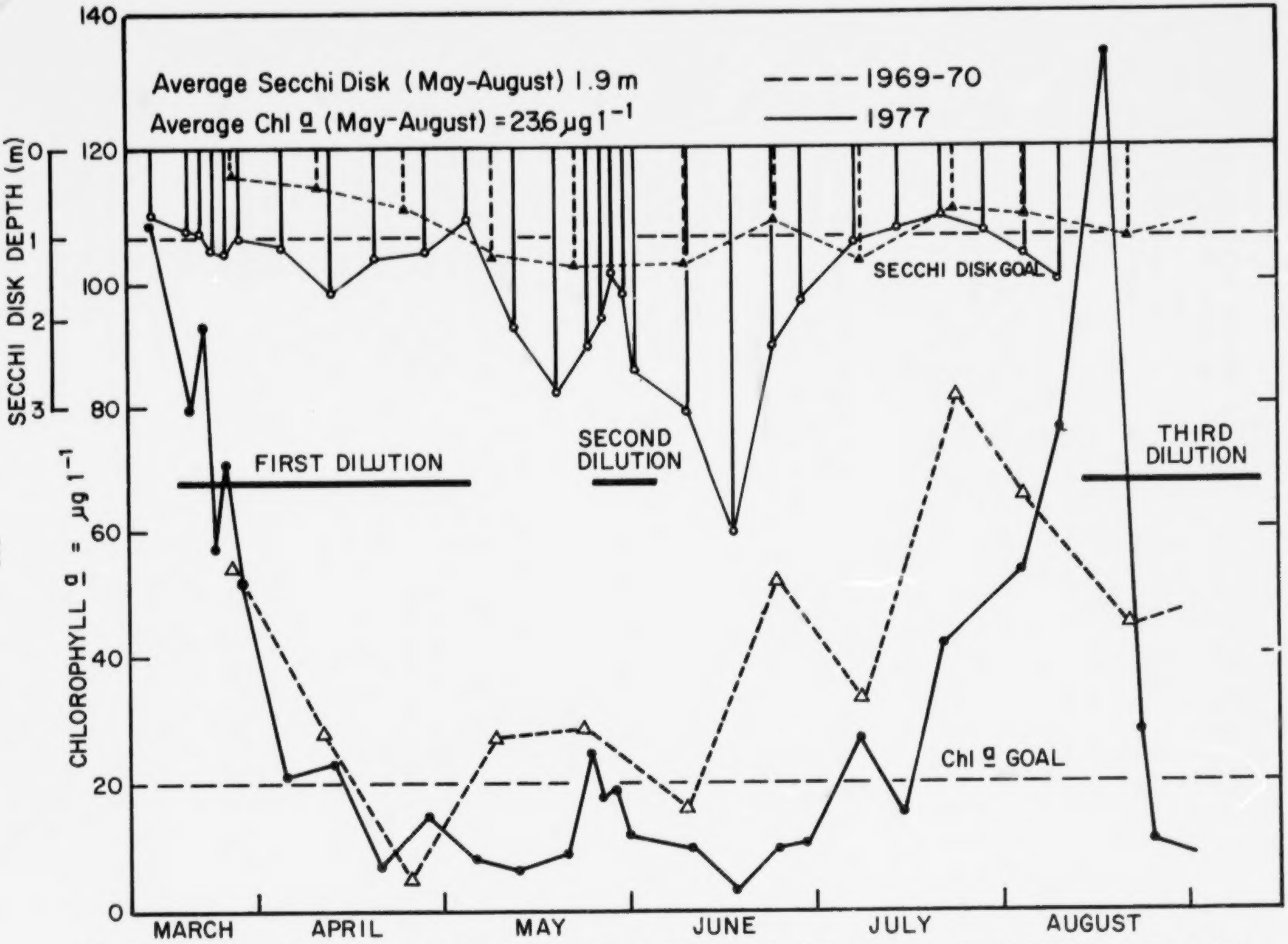


Figure 9. Chlorophyll a from horizontal transects and Secchi disk measurements at Station 9, Moses Lake, South of the I-90 bridge during 1969-70 and 1977.

during the addition of dilution water, the variability also indicates that other factors affected the levels. The P levels appeared to increase at the beginning of the first dilution water addition. That was no doublt a result of the detritus and debris that had accumulated in the wastewater and was washed into the lake with the introduction of the first quantity of water. A few days after initial addition the P level dropped and remained relatively low, at least to around the 50 μg l^{-1} goal, for at least the duration of the dilution period and for two to three weeks after termination of the addition. That was particularly true in Parker Horn, north of the I-90 bridge.

Clearly, the P level increased markedly during July and early August when dilution water was not entering the lake. The concentrations attained were commonly greater than 100 μg l^{-1}, and on one occasion in Parker Horn nearly 200 μg l^{-1} was attained. Since the concentration in Crab Creek averaged considerably less than that, the source of P that raised the levels so high must be internal.

Total N appeared to behave in a slightly more conservative manner than did P. During the first dilution period N was reduced and remained low and more stable than P. Of course, it increased during the non-dilution period in July and early August similar to P, but the levels were not closely correlated with P.

Figures 8 and 9 show the seasonal comparison of chl *a* and Secchi disk depth between 1969-70 and 1977. While the dilution water addition greatly reduced chl *a* during the first period, the level was not below the 1969-70 level. However, Secchi depth was considerably greater. This may have been due to the fact that normal , rather turbid, runoff in the springs of 1969-70 kept chl *a* down due to a high exchange rate, but also held the clarity low because of non-algal turbidity.

For the remainder of the summer, however, the chl *a* level and Secchi depth were much improved over those in 1969-70. Secchi depth reached a maximum of 15 feet in the lower lake in mid June coincident with the minimum chl *a* content. The second dilution water addition, although the lowest rate of exchange (0.09^{-1} day), effectively reduced chl *a* and increased Secchi depth both north and south of the bridge. In fact, that observation is one of the more surprising ones in the study. In spite of P remaining relatively high at that time (well above 50 μg l^{-1} in the lower lake), chl *a* decreased promptly with the addition of the dilution water and remained below 20 μg l^{-1} for over a month.

During late July and early August, about 1.5 months after cessation of the second dilution, chl *a* increased rapidly to a level exceeding 100 μg l^{-1}. Chl *a* promptly dropped coincident with the start of the last dilution water influx.

The main point to be observed in the Figures 6 through 9 is that dilution water had a more pronounced and lasting effect on chl *a* and Secchi depth than it did on P and N content. This implies that large amounts of added dilution water, which are necessary to attain low P levels, were not necessary to effectively reduce and control chl *a* and improve water clarity. This suggests one of at least two mechanisms that could be operating. Either sufficient time was not available for the algae to build a biomass in proportion with the higher-than-predicted P content caused by non-inflow (internal) sources of P, or that some factor(s) in the dilution water discouraged the growth of blue-green algae, which resulted in their biomass decrease in proportion to the rate of water exchange.

PREDICTED VERSUS OBSERVED EFFECTS

To illustrate this point further, the observed changes in the pertinent variables are compared with predicted values in Parker Horn based on the previously mentioned equation of continuity. First, specific conductance was used as a conservative "element" and Figure 10 shows that reductions in lake levels were reduced largely in proportion to the input of dilution water. Total N was less consistent (Figure 11), but was affected more consistently than Total P (Figure 12) as was indicated previously in Figures 6 and 7. In Figure 12, the large peak of 100 µg l^{-1} after the beginning of the first dilution period had already been ascribed to debris from the wasteway canal washing into the lake. However, the other, seemingly anomalous peaks during the second and third dilution periods, were coincident with windy weather during sampling. In shallow unstratified lakes, the effect of vertical resuspension of settled matter through wind-driven mixing is probably an effective internal source of P. This then appears to be a strong inhibiting factor to the effective decrease in lake P content by the addition of low nutrient dilution water in a shallow lake.

As it turns out, however, the reduction of P does not appear to be a prerequisite for effective control of algae in Moses Lake by the addition of dilution water. As shown in Figure 13, the reduction of chl a in the lake follows the predicted decline almost precisely in the manner of a conservative property. That is particularly true for the last dilution period. If the algae were growing, then the biomass level, which is the difference between growth and loss, should have remained at higher levels than those predicted. This was actually true during the first period, but not the second and third. Even the low rate of water exchange during the second period (0.09/day) was adequate to cause total washout of biomass.

The algal crop that ceased growth and washed out during the third dilution period was nearly 100 percent bluegreens, mostly Aphanizomenon. As shown in Figure 14, the large biomass at the start of the dilution period was practically 100 percent bluegreen algae. The decrease in chl a as a result of washout was nearly proportional to that of bluegreens. Also observable is that the rapid decrease in bluegreen algae was accompanied by an increase in diatoms particularly, but also some green algae, to replace the bluegreens. Diatoms increased from 2,000 cells ml^{-1} on 8/10 to 55,000 on 9/29. Green algae increased from 2,000 ml^{-1} to over 20,000 ml^{-1} and decreased again to 8,000 ml^{-1} by 9/29. By October, one month after the cessation of dilution, bluegreens had reattained their dominance (82 percent).

Clearly then the presence of Columbia River dilution water greatly reduces and even stops the growth of bluegreen algae in the lake. This was shown in experiments performed by Buckley (1967) in which the quantity of

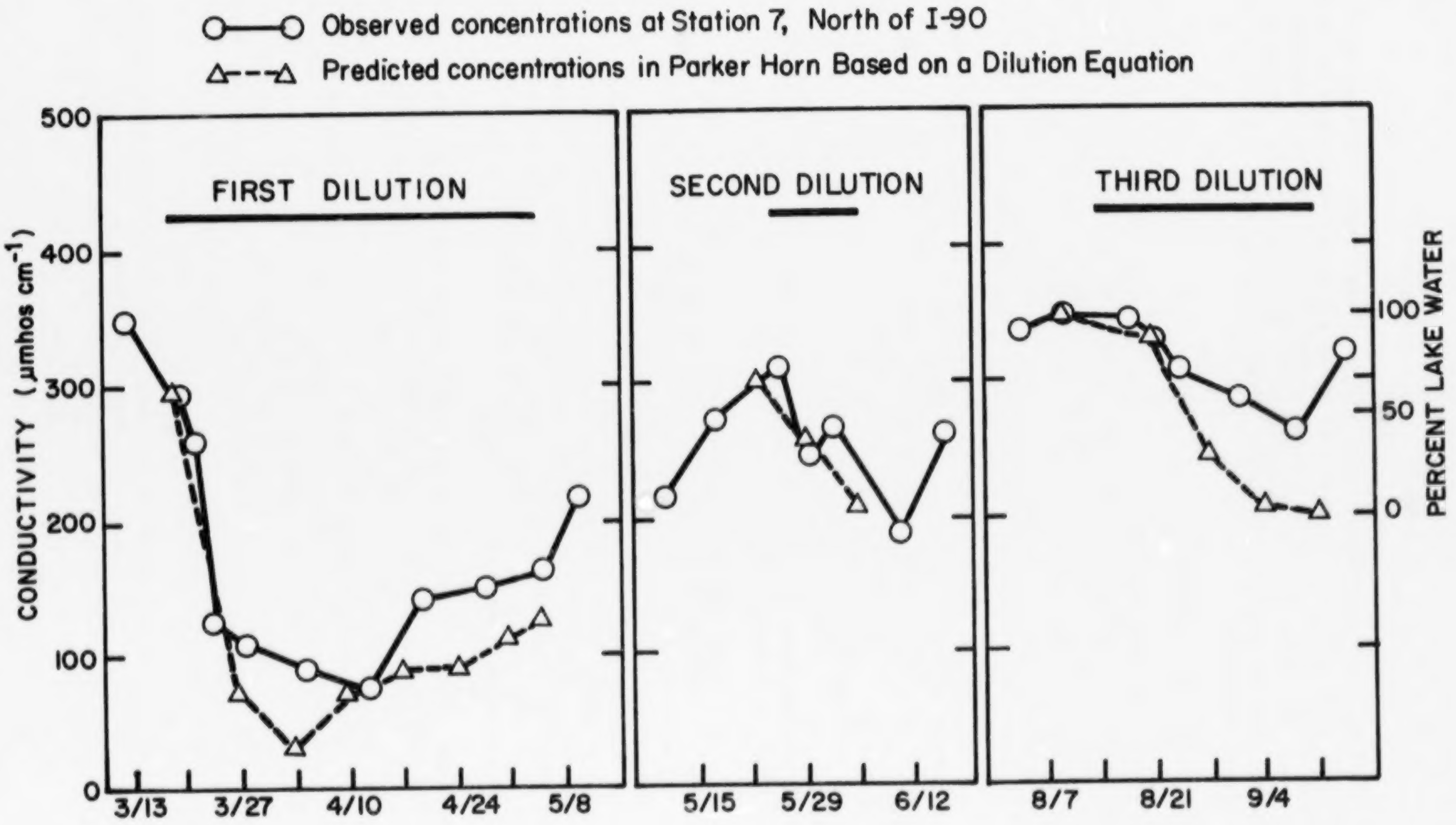


Figure 10. Predicted and observed specific conditions in Parker Horn, 1977.

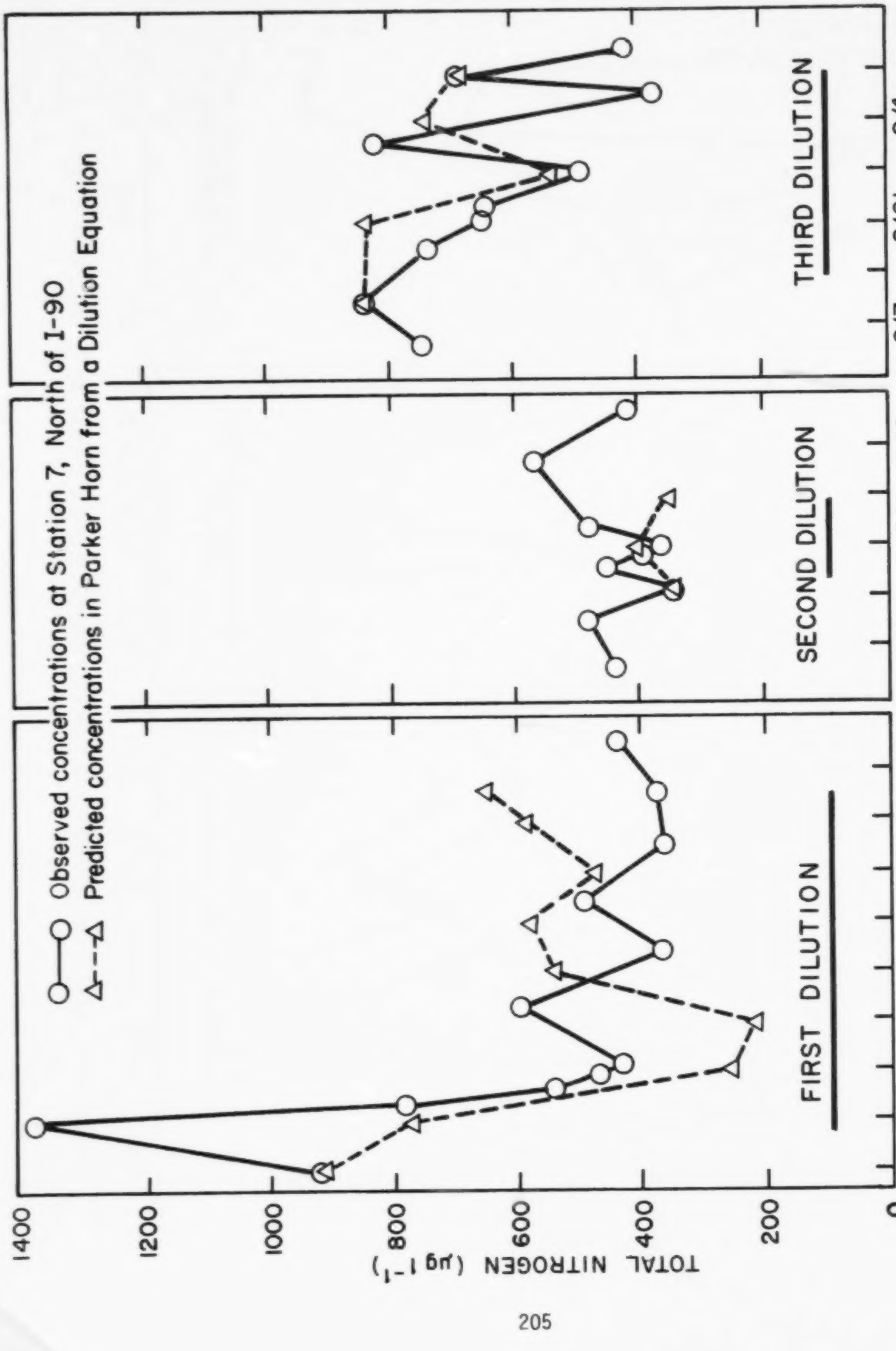


Figure 11. Predicted and observed Total Nitrogen concentration in Parker Horn, 1977.

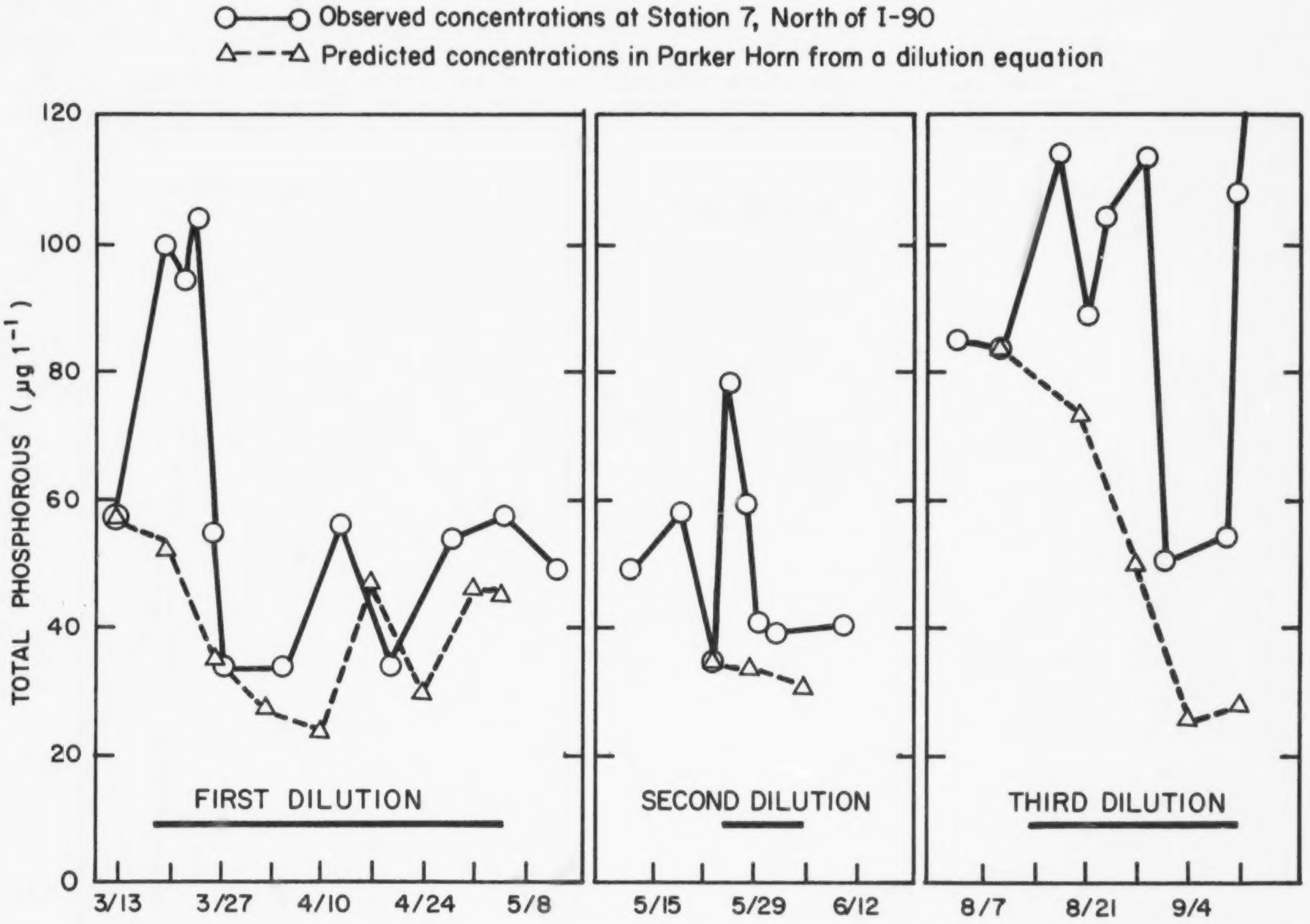


Figure 12. Predicted and observed Total Phosphorus concentrations in Parker Horn, 1977.

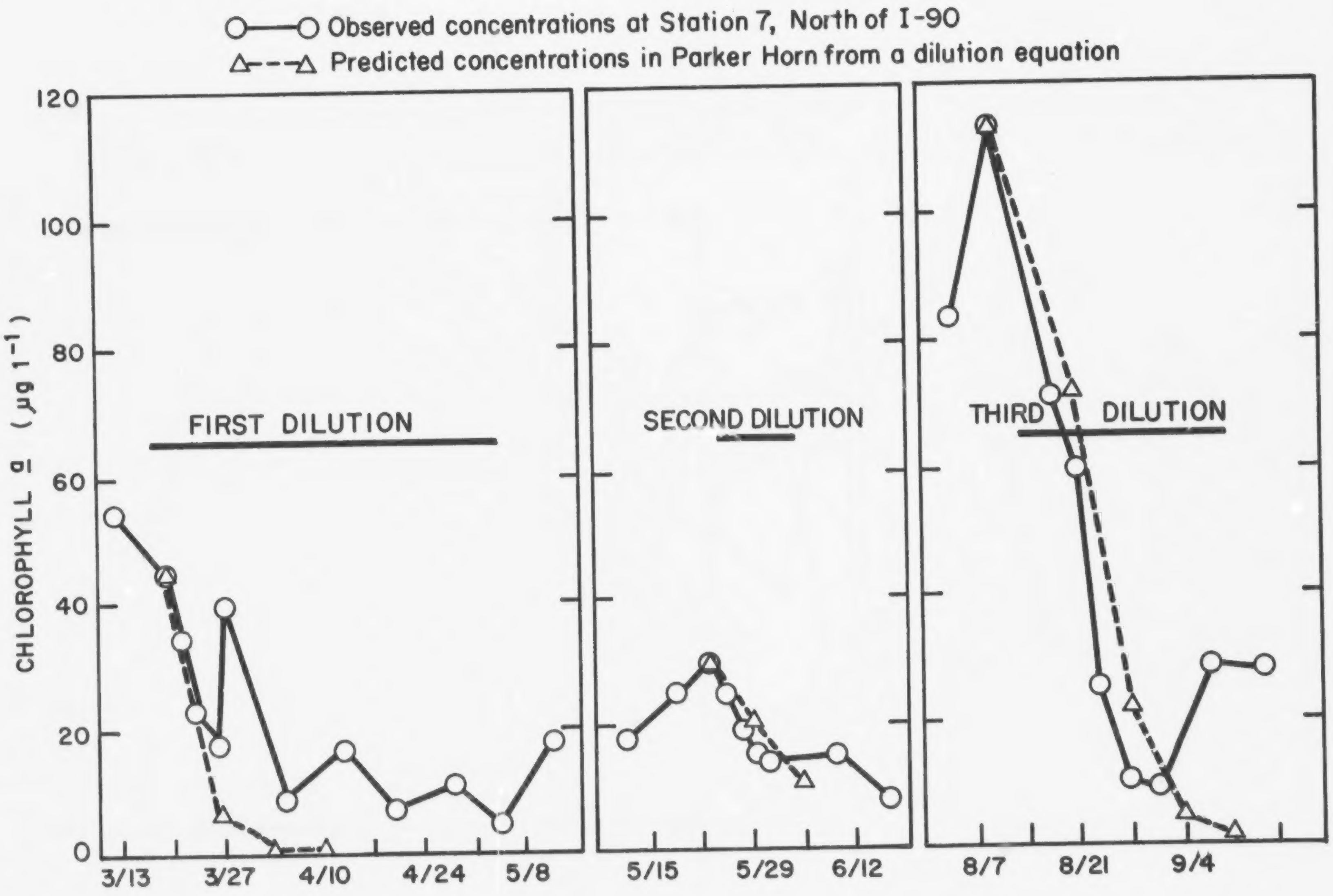


Figure 13. Predicted and observed Chlorophyll a concentrations in Parker Horn, 1977.

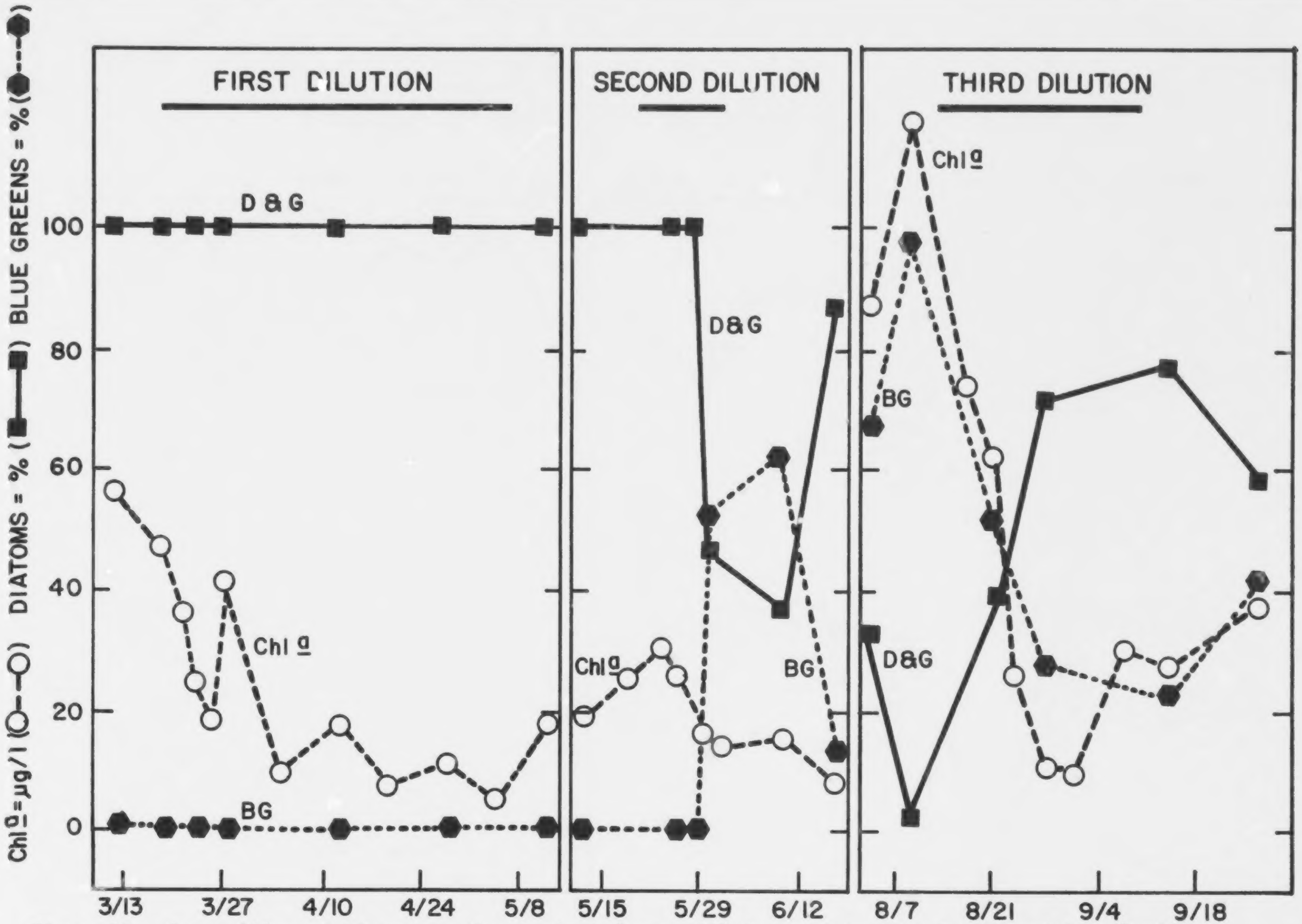


Figure 14. Composition of bluegreen algae, where "one cell" is equal to 40 m length and 400 m^2 of area for filaments and colonies, respectively, and diatoms + greens together, compared to chlorophyll a and dilution period during 1977.

bluegreens, after three weeks of exposure in 0.5 m deep plastic bags, decreased in direct proportion to the amount of dilution water added (Figure 15). This was thought to be caused by the reduction in P content, which is also indicated in Figure 15. At concentrations below about 50 µh l^{-1} P it appeared that little bluegreen algae was produced. This was also tied to the growth rate as shown in Figure 16. As the percent lake water decreased to around 25 percent or less, growth rate approached zero. At lower levels the bluegreens apparently died. Buckley also observed that as bluegreen growth decreased with increased amounts of Columbia River water the growth of diatoms increased, which is what actually occurred in the lake in 1977 and was particularly evident during the August dilution. Earlier in the season, the lake contained so much relatively "low" nutrient Columbia River water that diatoms (and some green algae) completely dominated the plankton.

The percent of lake water remaining at which complete cessation of growth of bluegreens occurs probably varies seasonally. Growth of diatoms did not appear to stop during the first dilution period, as evidenced by the larger than predicted biomass remaining (Figure 13) even though exchange rate was highest at 0.25/day. In August, growth of bluegreens was apparently stopped at a water exchange rate of 0.15/day. The percent lake water that was reached during the last dilution was about 50 percent, based on conductivity. However, washout was occurring earlier when the percent lake water was about 75. The theoretical level to which the percent lake water should have been reduced to is about 0 percent (see predicted curve in Figure 10). It appears that a rate of input of dilution water of about 20 m^3 sec^{-1} (700 cfs), that would theoretically result in a 0 percent lake water in about 2 weeks, is more than enough to attain adequate control of bluegreen algae.

While the experimental dilution rates were not low enough to observe an optimum input for control, there is room to speculate that an input of about 2 to 3 times the flow of Crab Creek should be adequate to prevent a dominance by bluegreens and maintain relatively low levels of chl a. Such an input of 2 to 3 times Crab Creek would be between 100 and 200 cfs (2.8 to 5.7 m^3 sec^{-1}) of Columbia River water depending on the base flow of Crab Creek.

SUMMARY

1. The addition of Columbia River water to Parker Horn and the lower portion of Moses Lake in spring-summer of 1977 greatly improved lake quality. The effect occurred throughout the lower lake as well as in Parker Horn. Chlorophyll and phosphorus were greatly reduced and water clarity increased. The improvements, with respect to the goals previously set, were 83, 52-67 and 67-180 percent for chl a, P and water clarity, respectively.

2. Chlorophyll a was effectively reduced and Secchi depth increased by the physical effect of washout at water exchange rates as low as 9 percent per day and reduction in residual lake water to about 50 percent.

3. The effective washout of bluegreen algae succeeded in an effective reduction in and further control of biomass because the presence of dilution water apparently inhibits their growth. This was demonstrated in earlier experiments at lake water concentrations between 25 and 50 percent. This

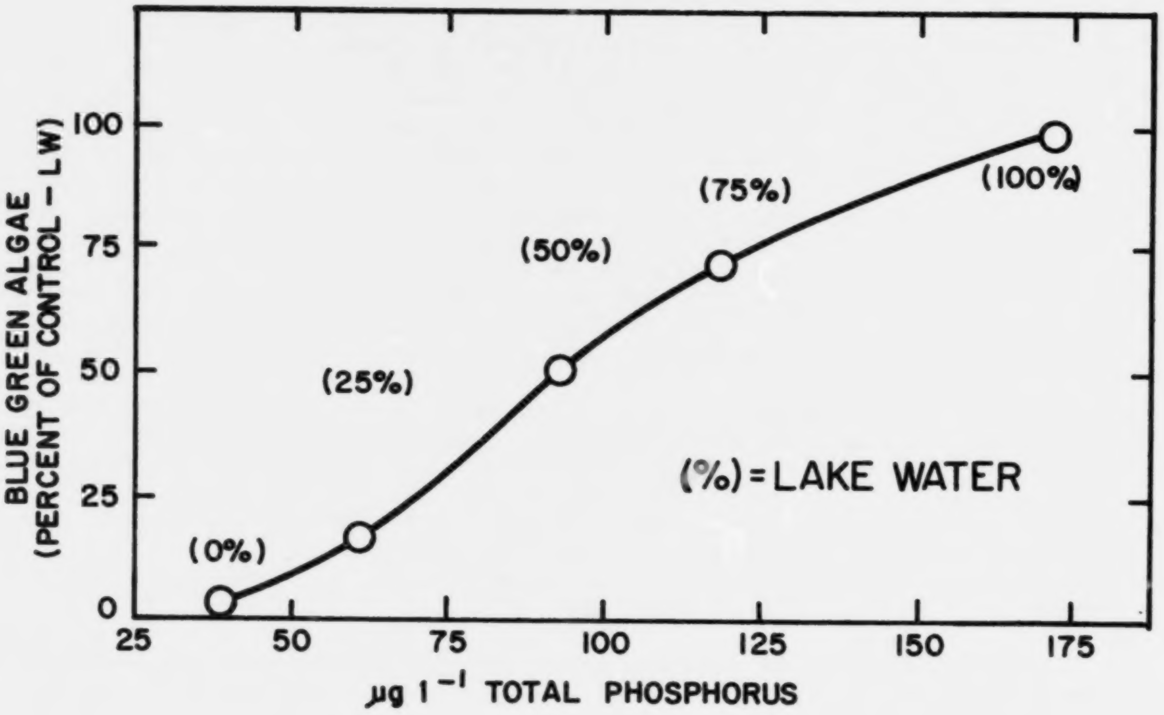


Figure 15. Maximum biomass of bluegreen algae in 100 l, 0.5 m deep plastic bags supplied with Moses Lake water diluted with Columbia River water to provide the given total P concentrations. The test water was renewed at 10% per day with the appropriate test. Moses Lake water and Columbia River water ratio to simulate continued flow of the mixture through the lake section, and lasted for 3 weeks.

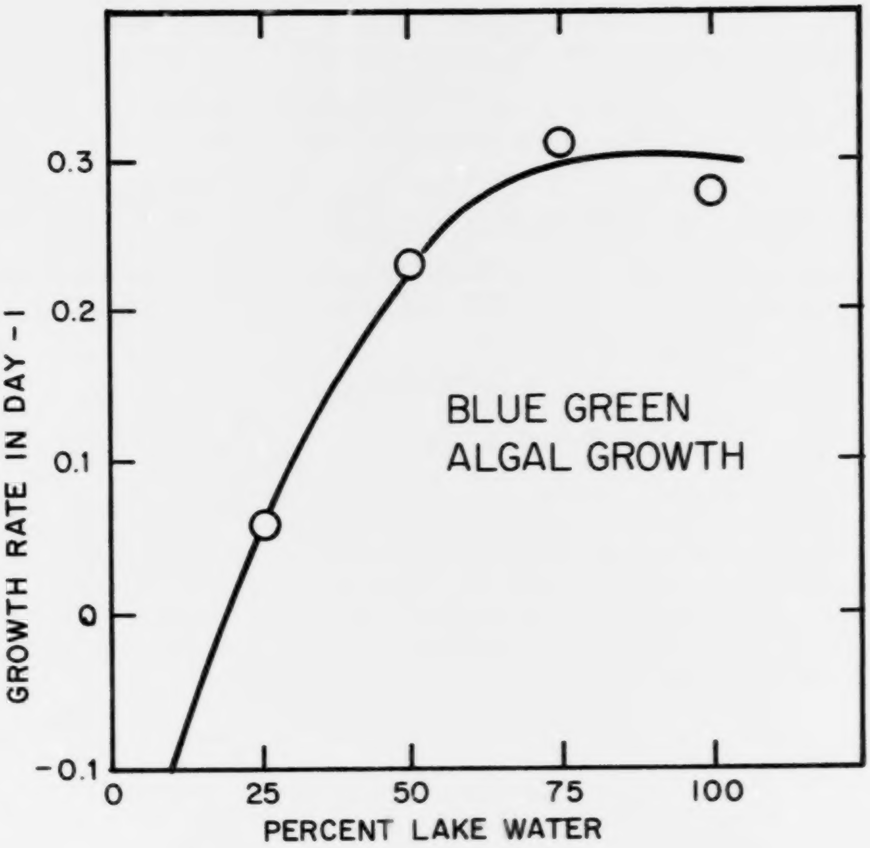


Figure 16. Growth rate of bluegreen algae exposed to various concentrations of lake water diluted with Columbia River waters with an exchange rate for the culture medium of 10% per day.

cessation of growth and washout of cells was observed during the third dilution period in August at a lake water concentration of about 50 percent based on specified conductance.

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DETAILED EVALUATION OF THE LONG LAKE IMPROVEMENT PROJECT

by

R. V. Blomquist and W. Wood*

My presentation will be on the Long Lake Improvement Project. The Detailed Evaluation Grant on this project has been awarded within the last month, and as a result, no work has actually begun on the Detailed Evaluation.

Today I would like to describe to you some of the rationale and the history behind the Long Lake Improvement Project itself, some of the reasons for its existence, and some of the goals of this project.

Long Lake is located in the city of New Brighton, a suburb directly north of the Minneapolis/St. Paul, Twin City area. New Brighton was historically a small agricultural community, near a large city, which has in the last fifteen years become engulfed by the spreading suburbs of the Twin City metropolitan area. New Brighton is a bedroom community of about 25,000 people, the majority of these people are employed in the downtown area.

Long Lake is the focal point of the Rice Creek Watershed District. Watershed districts may be a level of government organization which some of you are not familar with, but which are common in Minnesota. Watershed districts are established by order of the Minnesota Water Resources Board, acting under the authority of the Minnesota Statutes. One of the main purposes of the watershed district is to deal with matters which cross county and municipal boundaries. The affairs of the district are administered by a board of managers appointed by the County Commissioners of the affected counties. In this case, there are 5 district managers; two of which are appointed by Ramsey County, two from Anoka County and one from Washington County.

The Rice Creek Watershed encompasses 201 square miles which drain into Rice Creek and eventually into the Mississippi River. Long Lake is located at the focal point of the watershed and receives surface drainage from 195 square miles which contain rural areas, located in the north; and residential, commercial and industrially developed areas in the south. The lake itself has a surface area of approximately 200 acres and is divided into two basins, a north basin containing about 75 acres and a southern basin containing about 125 acres. The mean lake depth is 12 feet. The maximum depth is 35 feet. Rice Creek enters the lake on the northeast corner and leaves the lake on the

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* National Biocentric, Inc., 2233 Hamline Avenue, North, St. Paul, MN 55113

northwest corner. Other areas drain into the lake, entering it from the south basin.

Long Lake has experienced problems of excessive sedimentation in the northern basin, an overall degradation in water quality, and occasional local flooding in the area.

The Long Lake improvement project encompasses not only Long Lake, but involves a series of restoration measures undertaken to improve water quality in a chain of lakes to the south. In addition to Long Lake, other lakes that will benefit from this project include Lake Johanna, Lake Josephine, Valentine Lake, Pike Lake, Jones Lake and other small lakes in the immediate vicinity. The Rice Creek Watershed District submitted the grant application and was subsequently awarded a Lake Restoration Grant to conduct the improvements on the Long Lake chain of lakes.

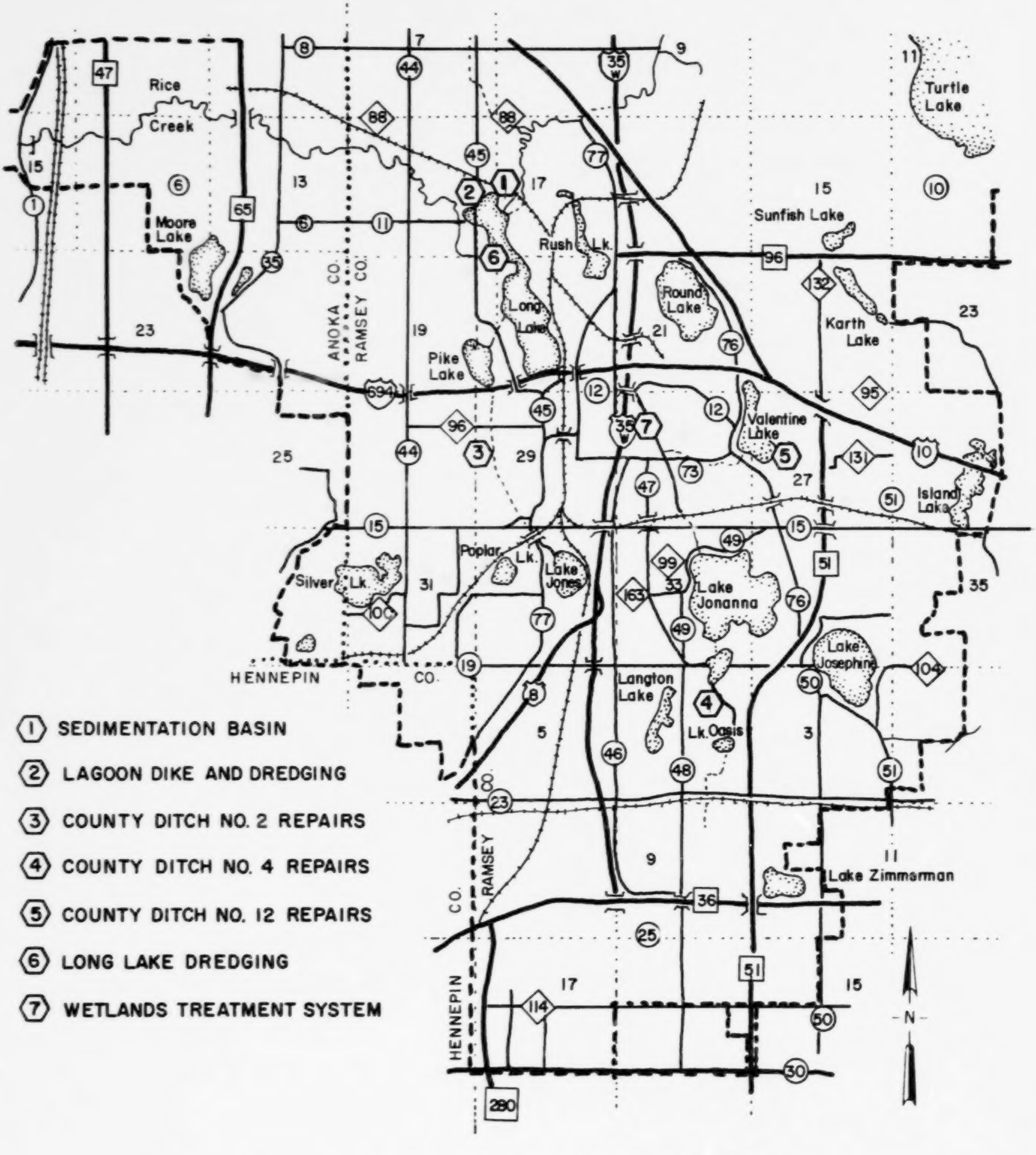
The improvements can be divided into four broad categories (Figure 1). First, a sedimentation basin will be installed in Rice Creek before it enters Long Lake. Second, channel repairs and upstream improvement to control erosion are to be undertaken. The various improvement projects are spread throughout the area of the watershed. It should be noted that the upstream improvements and channel repairs are proposed in the more heavily urbanized areas of the watershed. The third category involves the dredging of a portion of Long Lake which has filled in as a result of sedimentation which has occurred over the last decades. Fourth, is a wetland treatment system which will be installed in an area which receives a large amount of runoff, and will allow for treatment before the water enters Long Lake.

The transport and deposition of sediments in the Long Lake system has been studied in an effort to determine the quantity as well as the sources of the sediments. Three sources, Rice Creek, Pike Lake, and Lake Johanna, are the major tributaries to Long Lake and contribute the majority of sediments being deposited in the lake. The sediments result from sheet erosion in open space areas and channel and stream bank erosion in other locations, which are then transported to the lake.

Of the three sources, Rice Creek and its associated upstream watershed, constitute the largest single source of sediment deposition. The sediment load via Rice Creek is estimated to be 2,000 tons per year. Approximately 200 tons per year are estimated to result from Pike Lake and an estimated 500 tons per year from Lake Johanna.

Nutrient balance considerations for Long Lake indicate that 13,000 pounds of phosphorus enter the lake annually, of this amount, 5000 pounds are deposited as bottom sediments or utilized by aquatic plants and 8,000 pounds are discharged via the lake outlet. Rice Creek accounts for approximately 60% of both the water and the nutrients which enter Long Lake annually.

At the inlet of Rice Creek to Long Lake, a delta has been forming as the result of deposition of sediments over the years. Currently, in the spring of the year, water flows down the creek and into the lagoon before moving out into the lake. The improvement project calls for dredging the inlet to Long Lake, and allowing the water to move directly into the lake.



① SEDIMENTATION BASIN
② LAGOON DIKE AND DREDGING
③ COUNTY DITCH NO. 2 REPAIRS
④ COUNTY DITCH NO. 4 REPAIRS
⑤ COUNTY DITCH NO. 12 REPAIRS
⑥ LONG LAKE DREDGING
⑦ WETLANDS TREATMENT SYSTEM
Rice Creek
Moore Lake
Turtle Lake
Sunfish Lake
Rush Lk.
Round Lake
Karth Lake
Long Lake
Pike Lake
Valentine Lake
Island Lake
Poplar Lk.
Lake Jones
Silver Lk.
Lake Jonanna
Lake Josephine
Langton Lake
Lk. Oasis
Lake Zimmerman
ANOKA CO.
RAMSEY CO.
HENNEPIN CO.
RAMSEY CO.
HENNEPIN
N

Figure 1. Long Lake chain of lakes improvements.

PHOSPHORUS BALANCE

Inlet	Lbs.	Outlet	Lbs.
Rainfall	23	Rice Creek	7,984
Direct R.O.	814		
Rice Creek	8,035		
Pike Lake	817		
Lake Johanna	3,615	Difference	5,320
	13,304		13,204

From: Grant Application, Long Lake Improvements. Rice Creek Watershed District, 1976.

WATER BALANCE

Inlet	Million Gallons Per Year	Outlet	Million Gallons Per Year
Rainfall	135	Rice Creek	10,405
Direct R.O.	154	Evaporation	180
Rice Creek	6,256		
Pike Lake	1,053		
Lake Johanna	2,987		
	10,585		10,585

From: Grant Application, Long Lake Improvements. Rice Creek Watershed District, 1976.

Immediately upstream is the general area for construction of a sedimentation basin. It is anticipated that this basin will remove much of the sediments carried from the large rural portion of the watershed into the lake. The sediment basin is designed to accommodate the three-year rainfall event and to have a five-year clean out interval. This represents a storage capacity of approximately 9,000 cubic yards.

The Pike Lake sub-watershed, and particularly County Ditch #2, has significant erosion problems. It is proposed to coordinate municipal park development projects with the lake improvement techniques in this area to maximize efficiency.

The Lake Johanna portion of the watershed carries water from Lake Josephine, Johanna, and from Island, Valentine, and into Long Lake. Starting in the southernmost portion of the watershed, Lake Zimmerman receives runoff from freeways and urban development, both residential and commercial; there's even a golf course over on the right-hand side of the lake. Water from this southern portion of the watershed drains north into Lake Johanna. The southern area of the watershed, that which is drained by Lake Zimmerman, is primarily residential; whereas that drained by County Ditch 14 is primarily commercial and industrial. Important features of this area of the watershed are the trucking operations and shopping centers with their large parking lots, which present problems from the standpoint of runoff. The ditches which drain the parking areas and shopping centers from this portion of the watershed are very large in order to handle the runoff. When these ditches fill immediately after a heavy rain, large volumes of water move through these ditches and carry along debris.

Water also enters Lake Johanna at its southeasterly corner. There is a beach located on Lake Johanna that receives quite a bit of usage. The outlet from Lake Johanna is not large enough to effectively handle the outflow and can sometimes act as a dam after heavy rains. Part of the improvement project calls for improving the outlet and increasing its ability to handle large volumes of rain without creating flooding conditions.

Island Lake is located in the northern half of the Lake Johanna subwatershed. Water from this area moves through Island Lake, Valentine Lake, and finally joins up with the water from Lake Johanna. Island Lake is surrounded by primarily residential areas; however, it does receive some drainage from industrial office-type facilities.

The area between Lake Johanna, Valentine Lake, and Long Lake is proposed for the construction of the wetland filter system. There will be some biological uptake, some entrapment, and some microbial activity which would remove some nutrients from the water before it enters Long Lake.

That pretty well summarizes the lake improvement project. The improvement project has not been designed by our firm, or people under my supervision.

The detailed evaluation program on Long Lake has two aims: first , to evaluate the specific treatment projects, that is the sedimentation basin, the wetland filters, the channel repairs and the dredging, to determine whether they are meeting their design specifications; and, the second objective will be to evaluate the water quality in Long Lake and also in the other lakes in the system, to determine whether or not the lake restoration techniques have had any effect on water quality.

Because of the position of Long Lake with respect to the entire watershed, the limnological evaluation will include not only Long Lake itself but other lakes in the watershed. However, the primary effort will be on Long Lake. As stated earlier, the actual evaluation has not begun at this time; however, we anticipate collecting some of our first samples through the ice within the next two weeks. There are still thick layers of ice on the lakes in Minnesota. In fact, when I left on Monday, only one day had been recorded when the temperature had been above freezing since December 19th.

EVALUATION OF LAKE RESTORATION PROCEDURES

1. Sediment Basin
    - Upstream versus Downstream
        - Hydrologic
        - Sediment
        - Nutrient
    - Continuous Hydrologic
    - Regular Water Quality (Lake and Solids)
    - Periodic Storm Event

2. Erosion Control/Channel Improvements
    - Not Well Defined
    - Regular if Possible
    - Periodic Storm Events

3. Wetlands Filter
    - Upstream versus Downstream
        - Hydrologic
        - Sediment
        - Nutrient
    - Regular
    - Periodic Storm Event

4. Dredging (Not Well Defined)
    - Lake Depth
    - Nutrient Removal
    - Sediment Removal

Since improvement of water quality is one of the primary objectives of the lake improvement project, the limnological investigation will play an important role in the overall detailed evaluation. The success of this evaluation is based on both the base line data and the data collected in years following the implementation of the improvement technique. At this point, I would not like to comment extensively on the potential problems in scheduling except to point out that the schedule for the resotration project has not been firmed up, and may, in fact, continue beyond the completion date for the detailed evaluation.

Our schedule for Long Lake calls for the initiation of water quality monitoring in the spring of 1978. As I indicated, we will likely be in the field within the next two weeks to collect samples through the ice. At that time, we will also collect sediment cores and begin some of the dating evaluations of the sediments. Our program calls for monthly water quality sampling through the ice. Samples will be collected every three weeks from June through November, and on a bi-weekly basis from ice-out through May. Surface water and volume proportional composite samples will be collected and analyzed for the phosphorus complex, the nitrogen complex, chlorophyll a and alkalinity. On site field data, such as Secchi disk, dissolved oxygen profile, and conductivity profile, will be collected. Extensive data regarding time of sampling, climatological data, will also be recorded.

The biological analysis will be aimed at trying to collect data which will allow for a complete understanding of the relationships of the lake's aquatic ecosystem. Algal data will include species identification and enumeration. Primary productivity will be analyzed in the spring, summer and fall. Alkaline phosphatate activity will be evaluated throughout the year in an effort to assist in the interpretation of the relationships between many of the inlake factors. Analysis of the macrophyte population will be conducted twice per year to identify the species present, determine the standing crop, and estimate productivity. Zooplankton analysis will be conducted during the summer months to determine the species present and the population size. This information will allow for comparisons between algal abundance and zooplankton present. At this time, it is proposed to conduct some fishery analysis in Long Lake. This information should be useful in determining the overall aquatic relationships.

The hydrology of Long Lake is complicated by the fact that the lake consists of a north and south basin. The inflow from Rice Creek comprises the major hydrologic input and enters at the northeast corner of the north basin. The outflow is also from the north basin; however, there are also inlets to the south basin, and there is also interchange between the two basins. It is our aim to get a better handle on the hydrology of the Long Lake system. This should be useful in interpreting the limnological results and evaluating the effect of the lake restoration project.

In addition to the surface water entering and leaving Long Lake, we are proposing to evaluate the groundwater relationshps with respect to the lake. Based on existing data, we have been able to conclude that there is a complex groundwater system in the vicinity of Long Lake. Existing information indicates that Long Lake lies diagonally across a buried bedrock valley. We are proposing to drill observation wells and conduct a simulation analysis of the groundwater flow in the area.

Although the improvement project is aimed at removing external sources of nutrients and sediment, internal loading can be an important factor which could delay the response of a given lake as a result of decreased external loading. We are proposing to evaluate the direct internal loading from the sediment, as well as indirect loading as a result of bottom feeding fish or macrophytes, which may pump nutrients from the sediments into the water.

I would be happy to entertain any questions you may have. Once again I'd like to point out that we are at the organizational stages of our evaluation. However, based on what you have heard this morning, any comments particularly suggestions, would be appreciated. Working on this project with me are Mr. Will Wood, who is the head of our National Science Group at National Biocentric, and Dr. Joseph Shapiro, from the University of Minnesota.

A BCCIPA MODEL FOR WATER RESOURCE PROJECT EVALUATION*

by

Ben-chieh Liu**

INTRODUCTION

Enactment of the comprehensive Federal Water Pollution Control Act (FWPCA) Amendments of 1972 (PL92-500) culminated nearly 3 years of executive and Congressional deliberations aimed at strengthening the clean water program. Consistent with the President's proposed legislation, it extended federal-state regulation to all navigable waters, authorized stringent federal standards or prohibitions for toxic discharges, strengthened and streamlined federal enforcement procedures, etc. The Act required states to develop a comprehensive and continuing planning process for water quality management. Plans had to include not only the point source controls but also controls for diffuse land runoff and other nonpoint sources. Beginning in 1975, the states had to submit annual reports to EPA that inventory all point sources of pollution, assess existing and anticipated water quality, and propose programs for nonpoint source control.

Implementation of PL 92-500, Sections 314/104(h) (the Clean Lakes Program), is part of a major federal, state and local effort to clean up the nation's polluted lakes. Under this program, state and local governments are encouraged to classify lakes and then to design projects which will not only control the problem sources but also restore or improve the water quality of the lakes.

However, as pointed out by Peterson and Porcella (1977), most of the lake restoration techniques being employed under the Clean Lakes Program is not only necessary to determine the effectiveness of various techniques, or demonstration projects, which have been funded for cleaning lakes, but it is also crucial to compare the relative effectiveness of these techniques as applied to different lakes, since there will be numerous future applications. Because they are publicly funded, the relative effectiveness of the clean lakes programs must be evaluated not only from the standpoint of limnological changes, but also from various impacts on social and economic welfare. Since resources are finite and environmental protection or pollution control is costly, it is essential to ascertain that the last unit of clean lake budget imposes no additional costs greater than the additional benefits which can be generated by the project.

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The primary objective of this paper is to develop a comprehensive impact assessment model for lake restoration project evaluation. It is hoped that through U.S. case studies, not only will the usefulness of the model be demonstrated, but also insight into the differences between social and private considerations, joined and separated projects, current and future alternatives, etc., will be furthered, and the conflicts between efficiency in resources allocation and equity in income redistribution will be highlighted.

THE BCCIPA MODEL

To evaluate the effectiveness of various lake restoration projects funded under the Clean Lakes Program it is necessary not only to determine the level of water quality improvement through a limnological study, but also to compare the relative cost effectiveness of the techniques employed in different lakes. This latter requirement, in essence, is to conduct a thorough benefit/cost analysis by which various private and social impacts can be compiled, and the program effectiveness evaluated. To accomplish this objective, an integrated Benefit/Cost Cross-Impact Probabilistic Approach Model (BCCIPA) for measuring direct, indirect and induced project impacts is presented below.

DIRECT IMPACT ASSESSMENT

The proposed BCCIPA is a synthesis of existing proven techniques for project impact effectiveness assessment. Instead of using any single technique, it adopts a systems approach consisting of several conventionally employed techniques; including both quantitative methods, such as time and extrapolation (export-base, shift-share, gross product accounts, etc). and the regression and simulation models (I/O), dynamic simultaneous equations, etc.), and the qualitative methods, such as risk analysis, the Delphi approach, multidimensional scaling, scenario development, cross-impact analysis, etc.

The BCCIPA proposed in this study, as can be seen from Figure 1, far from being a single technique, is a system of methods of measuring socioeconomic and environmental impacts. The entire scope of assessment is described under two general categories. The tangible direct and indirect aspects will be quantitatively identified and measured through the broad benefit/cost scheme of computation; whereas the intangible indirect and induced elements will be delineated, and evaluated, under the multidimensional scaling probabilistic approach. Since the nature of benefits/costs to be identified and measured differs between tangibles and intangibles, the specific techniques proposed are also different.

Following Liu (1977) and others, the benefit (B) cost (C) model, in symbolic form, appears as follows:

$$V = \sum_{i=1,\, j} \int_{t=1,\, n} \left\{ [B_i(t) - C_i(t)] \,/\, (1 + r)^t] \right\} dt_i ,$$

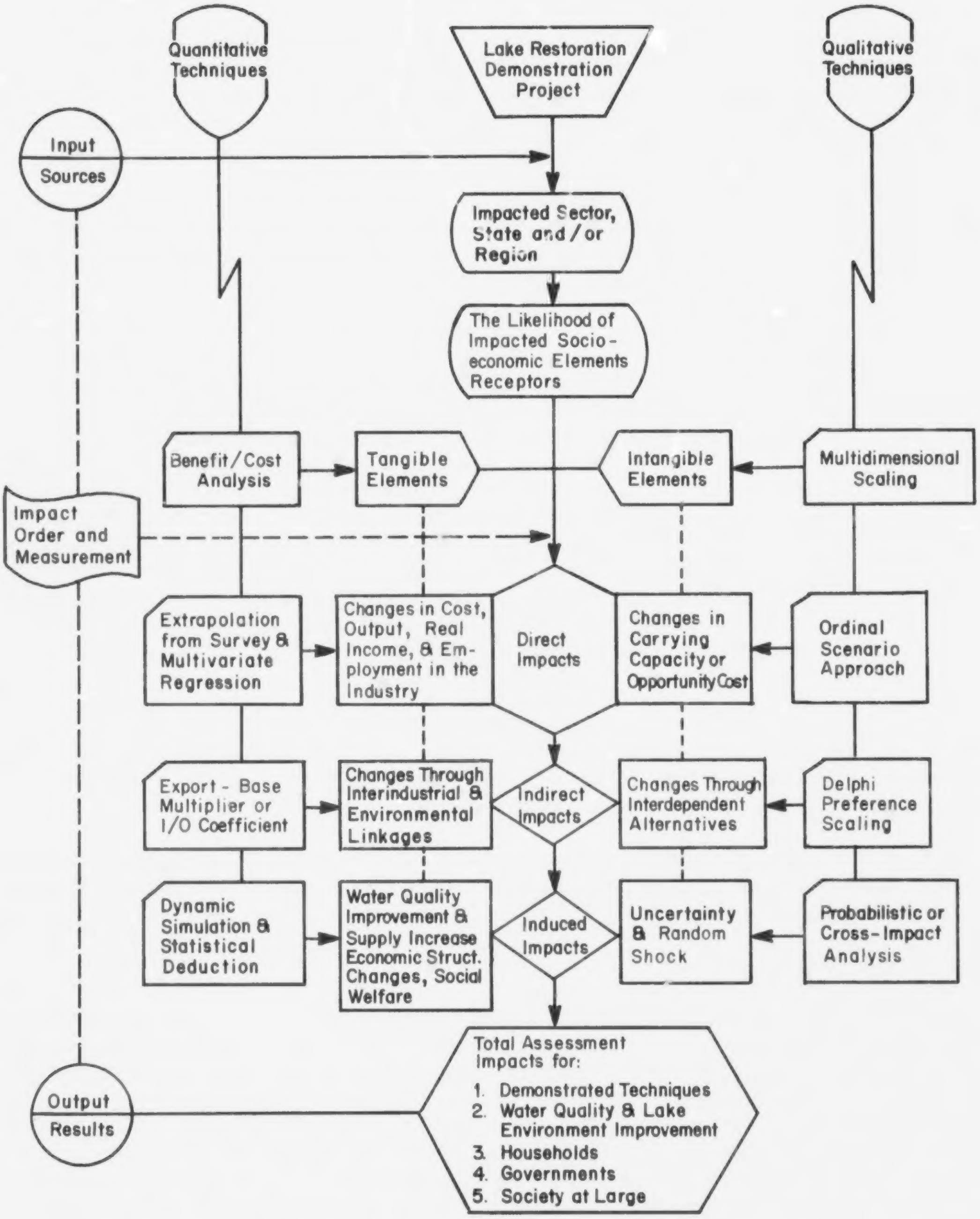
Quantitative Techniques
Lake Restoration Demonstration Project
Qualitative Techniques
Input
Sources
Impacted Sector, State and / or Region
The Likelihood of Impacted Socio-economic Elements Receptors
Benefit / Cost Analysis
Tangible Elements
Intangible Elements
Multidimensional Scaling
Impact Order and Measurement
Extrapolation from Survey & Multivariate Regression
Changes in Cost, Output, Real Income, & Employment in the Industry
Direct Impacts
Changes in Carrying Capacity or Opportunity Cost
Ordinal Scenario Approach
Export - Base Multiplier or I/O Coefficient
Changes Through Interindustrial & Environmental Linkages
Indirect Impacts
Changes Through Interdependent Alternatives
Delphi Preference Scaling
Dynamic Simulation & Statistical Deduction
Water Quality Improvement & Supply Increase Economic Struct. Changes, Social Welfare
Induced Impacts
Uncertainty & Random Shock
Probabilistic or Cross-Impact Analysis
Total Assessment Impacts for:
1. Demonstrated Techniques
2. Water Quality & Lake Environment Improvement
3. Households
4. Governments
5. Society at Large
Output
Results

Figure 1. Structure of the BCCIPA Model.

where i, t and r are, respectively, the ith type of benefit and cost quantified in time period t, which are weighted by the social rate of discount (r) to yield the net present value (V) of the project over its life span.

The baseline extrapolation technique will be employed to measure the direct improvement in water quality, recreational values brought about by the investment, and changes in basic economic variables, such as growth in employment and/or real income per capita. The ordinal-scaled scenario technique will be utilized to identify direct, intangible impacts, such as changes in carrying capacity, opportunity costs, and the aesthetic values. For instance, not only will the clean lake investment program have a direct economic impact on employment in that a number of new jobs may be created, but also the resulting high water quality will increase the recreational usage and aesthetic values of the lake as additional social benefits.

INDIRECT IMPACT ASSESSMENT

Indirect or second-order impacts arise when the direct impacts are viewed in concert with the environments within which the direct impacts take place. For instance, the changes in water resource capital investment policy directly affect the regional carrying capacity of water supply and, hence, the utilization and performance of various related public and private programs. The indirect impacts of this capital expenditure will also include the "substitution" and/or "stimulation" effects on the regional economy in resource allocation and distribution. The export-base multiples derived from the Leontief's (1970) Input-Output model may be employed, as by Liu (1971) and others, to measure the indirect impacts on income or employment. The Leontief's I/O model appears in matrix form as the following:

$$(X - A X) = Y; \text{ or } (I - A) X = Y \text{ and } X = (I - A)^{-1} Y$$

where X, Y, and $(I - A)^{-1}$ are, respectively, the intermediate goods and services, the final demand for goods and services, and the multiplier itself; A is the technical input-output coefficient matrix.

The Delphi preference scaling technique or subjective judgment method will be used to assess the changes through interdependent alternatives classified as the intangibles.

INDUCED IMPACT ASSESSMENT

Tertiary or induced impacts are further repercussions entirely within the physical and institutional environments of the first- and second-order changes and result from, but are not directly associated with, the direct impacts. They may occur as intended or be concomitant responses to the indirect impacts. The induced impacts of any environmental policy and/or program, like others, can also be differentiated according to the time lapse in which each event occurs. In addition to spatial and subjective matters of varying sequential importance, direct impacts are generally observed immediately, indirect impacts are created later, and tertiary impacts are felt much later.

While the dynamic simulation developed by Gordon and Haywood (1968), Johnson (1970), Rochbert (1970), Bloom (1977), Mitchell _et al_. (1977) and others may be constructed to identify and generate information on induced impacts brought about by regional structure changes, the proposed probabilistic cross-impact analysis will be utilized to evaluate the uncertain elements of the intangible, social and private benefits or costs induced through all sorts of externalities, and to develop a weighting scheme for induced impact quantification. Ultimately, it is hoped that the output of this model will provide sufficient objective information essential to decisionmaking, especially when weighing project efficiency criteria against project equity considerations. The cross-impact probabilistic approach may be summarized as follows:

$$\hat{P}_j = \begin{cases} P_j + P_j\,[1 - P_j]\,[S_{ij}(t - t_i) / t]; \text{ for } t > t_i \\ P_j, \text{ for } t \leq t_i \end{cases}$$

$$W_j = E(V)_j \cdot \hat{P}_j$$

where P_j is the estimated probability of occurrence of j by time t; $\hat{P}_j$ is the revised probability of occurrence of j by time t_i after i occurred; S_{ij} is a measure of the strength and mode of the impact of i on j; $E(V)_j$ is the expected value or subjective importance assigned for j; and W_j is the impact weight sought for overall benefit-cost aggregation.

MODEL APPLICATION AND DEMONSTRATION PROJECT EVALUATION

The proposed model will be applied to the selected lakes where the technical restoration demonstration project has already been launched. A 3 year observation is designed as the period under study for project evaluation so that incremental impacts can be better understood and the dynamic sequential evaluation procedures as proposed in the model can be employed, adjusted, and finalized, together with the limnological studies conducted simultaneously.

Furthermore, the results will also be compared externally against those "control lakes" where no such demonstration project whatever has been implemented. Nonetheless, the control lakes have to be homogeneous, if not nearly identical in nature, in terms of eutrophic, econologic, pollution and other environmental conditions to the studied lakes.

Thus, the cost-effectiveness of the demonstration projects will be finally evaluated not only through internal changes over a period of 3 years but also through external comparisons to better assess the direct, indirect and induced impacts.

Although the proposed model is to be applied to water resource project evaluation in the United States, it is expected to be useful in other countries and for other public investment projects as well.

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EFFECT OF RESTORATION PROCEDURES UPON LIBERTY LAKE, FIRST STATUS REPORT

by

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INTRODUCTION

STUDY SITE BACKGROUND

Liberty Lake originated as the inundated remnant of a glacier dammed valley. The lake now occupies a basin of approximately 316.1 ha (781 acres). The watershed is relatively undisturbed and drains an area of 3,446 ha (13.3 sq mi). The major tributary, Liberty Creek, is usually very low in nutrient content, unless heavy precipitation results in water exchange with the marsh that parallels the last .8 km (½ mile) of creekbed toward the lake. Mean residence time of lake water is three years and during the usual quiescent summer period weak stratification will occur from mid-May to September. Water temperatures, however, vary less than 5C from bottom to surface. Mean depth is 7.0 m (23 ft) and maximum depth is 9.1 m (30 ft). Other morphological characteristics are listed in Table 1.

HISTORICAL BACKGROUND

By the turn of the century three large resorts were in operation around the lake. A railroad line to Liberty Lake completed in 1905 brought thousands of visitors. Boating, camping and open air dancing facilities were offered to the public. A 1914 photo (Kalez, 1972) shows about 3,000 bathers and picnickers in front of hotels and pavilions. With the advent of better roads and improved automobiles, just prior to World War I, many of these recreational activities shifted to Coeur d'Alene Lake. By the mid-fifties another upsurge in recreational activity occurred at the lake. One large resort and three smaller fishing and boat rental establishments now provide services. A large ranch (≅804 ha) at the southern end of the lake was recently acquired by Spokane County and designated a county park. In 1977, it attracted over 40,000 visitors (Angove, 1977). Much of the original resort property is now occupied by year around homesites. Residential, commercial development, and Department of Game facilities occupy 85% of the shoreline. The remaining portion exists as the county park, camping area, and wildlife preserve at the inlet area of the lake.

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TABLE 1. SELECTED LIBERTY LAKE CONSTITUENTS AND PHYSICAL CHARACTERISTICS, 1974-75.

Parameter	Description		
Stream Inflow & Residence Time	Total Annual $6.3 \times 10^6 m^3$	Mean Lake Residence Time ≅3 years	
Estimated Phosphorus in Inflowing Waters	Total Annual 263 kg	Mean Concentration .03 mg/l	
Estimated Nitrogen in Inflowing Waters	Total Annual 2763 kg	Mean Concentration .27 mg/l	
Mean Concentration Dissolved Phosphorus in Lake Water (.45 μm filtered)	Mid to Late Summer .001 to .004 mg/l	Fall .02 to .04 mg/l	
General Lake Characteristics	Total Alkalinity	pH	Hardness as $CaCO_3$
	14 - 26 mg/l	6.6-9.3	15 to 37
Physical Characteristics	Shoreline Configuration	Development of Volume	Mean Slope
	1.27	1.27	1.9%
	Volume	Surface area	Mean Depth
	$20.23 \times 10^6 m^3$	316.1 ha	7.0 m

Waste disposal practice has consisted of sump holes and septic tanks after the pit privy stage. Unfortunately, large portions of the relatively shallow soils (Spokane series) are underlain by bedrock at a depth of .5 to 2 m with a 4 to 70% slope toward the lake; in turn, many of the homesites overlay this area. Soil column migration tests performed by Gibbons _et_ _al_. (1975) utilizing radioactive phosphorus (^{32}P) indicated a possible movement of phosphorus (up to 8 cm during a 24/hr period) toward the lake. Nitrogen as nitrate could be expected to move much faster.

The first sewage collection system built in 1910 diverted about 40% of the residential wastes encroaching from the western shoreline. In 1966, the system was enlarged to contain about 50% of the wastes. However, tests conducted in 1974 by Futrell, Redford and Saxton (now Michael Kennedy Enginers) suggest considerable exfiltration.

Kemmerer _et_ _al_. (1924) made the first reported water quality investigation of the lake on July 31, 1911 during what he called "a bloom stage" of

158,200 algae per liter. His counts included both green and blue green algae. In contrast, our counts on July 26, 1977 were in excess of 5.5 x 10^6 per liter. Kemmerer's field data show oxygen levels at 5.8 mg/l in the surface layers and 4.1 mg/l near the bottom. Unfortunately no nutrient or trace metal data were taken.

PREVIOUS STUDIES

Residents around the lake had noticed and complained about increased algae growth in the lake since the late 1950's and 60's. By 1968 large masses of decaying blue green algae consisting primarily of Anabaena flos-aquae and Aphanizomenon flos-aquae were being deposited upon the beaches along with fragments of aquatic weeds. Members of the Property Owners Association contacted the Washington State University Environmental Engineering section in 1968 for assistance in identifying the algae problem. In 1971 we began a modest cooperative water quality sampling program with the property owners and lake ecology committee for one year. These studies were repeated in 1973. A water balance study was also completed (Orsborn, 1973) at that time. These data suggested that nutrient inflow from Liberty Creek was low, with the exception of waters flushed through the marsh to the southern end of the lake.

From the time of completion of the latter water quality study sponsored by the lake Property Owners Association, occasional sample collection and analyses were made by WSU Environmental Engineering. Based upon these additional data a proposal to the Washington State Department of Ecology was made through the State of Washington Water Research Center to further examine nutrient constituents of the waters, soils and sediments of the Liberty Lake basin. A second major effort was to be made to determine the feasibility of alleviating the massive algal blooms (by aluminum sulfate treatment) until long term solutions could be instituted. The proposal was approved for funding in the early summer of 1974. Matching funds from the College of Engineering were utilized in the early spring to obtain nutrient runoff data in cooperation with the lake Property Owners Association.

Several sediment cores were also driven in the lake at that time and algal bioassays conducted upon the spring runoff waters. These preliminary data suggested that nutrient influx into the lake by Liberty Creek could simply not provide the amount of nutrients necessary to support the massive amounts of algae and weeds. Laboratory studies of nutrient release from cored sediments indicated that while the lake bottom is definately a source, the aerobic conditions and limited stratification that predominates in the lake would somewhat limit its contribution. Under present conditions the cored sediments did reveal a considerable increase in nutrients in the top 15 cm of the core (in comparison to lower sediments). Dating by ^{137}Cs (Ritchie et al., 1973) established an estimated unconsolidated deposition rate of 15 mm per year. Metal analysis of the cores also indicated a considerable increase in several metals in the upper 15-20 cm layers. The increase in Zn, Pb, Mn, Cu, etc. corresponds in time with shoreline cultivation and with the practice of disposing of metallic solid wastes such as tin cans, buckets, wire and other debris in the lake.

This practice was apparently common to many of the lakes in the Spokane region until relatively recent times (10-20 years). While these solid waste practices have largely ceased, the need to dispose of sewage from increased permanent human populations around the lake has grown. As previously mentioned, the shoreline area is now 85% developed and all remaining open areas back from the lake have been purchased for residential development, with the exception of the county owned marsh at the southern end and the large Spokane county public park at the southeastern end of the lake. Public access to the lake is excellent with the Washington State Department of Game maintaining a large fishing and boating launch area at the northern end of the lake with parking and restroom facilities.

A smaller access area is located in the Wicomico Beach area. All of these public and private facilities have led to increased summertime residential populations as well as an estimated 90 to 100 thousand tourist visits per year. Lake outline, surface inflow-outflow and previous sample stations are shown in Figure 1.

It is believed that there are insufficient nutrients either in the lake water or in the non-bloom producing algae characteristic of the mid-summer period to account for the massive late summer blue green blooms. Whether the weeds deteriorate because of lower temperature or less light intensity is not known at this time, but the prodigious numbers of blue green algal cells appear to be the direct result of nutrient release from the weed beds. Solski (1962) has shown that 20 to 50% of the phosphorus content of macrophytes may be released within a few hours after death, and at least 65% of the remaining content over a longer period of time. Hutchinson (1957) also postulated the rapid decomposition of littoral vegetation as a possible phosphorus source feeding algal blooms. Recent studies at Liberty Lake by Kaufmann (1977) have further documented the weed-algae cycle. Kaufmann, while studying the growth of periphyton upon natural and artificial substrata at five lake stations, noted clouds of plankton appearing among and in the vicinity of deteriorating weed beds. He made cell counts in the weed areas and in the open waters and found at least one magnitude of difference, with greater numbers in the weed bed areas, during the late summer months.

Based upon the previously described data, it was decided to aim a large scale aluminum sulfate treatment at a time to intercept the fall nutrient release from the weed beds, as well as to reduce ambient levels of dissolved phosphorus before it could be incorporated into blue green algae. In October, 1974, 95.3 metric tons (105 T) aluminum sulfate were distributed by barge over a four day period (Funk et al., 1975, 1977). A moderately large Anabaena flos-aquae bloom (8000 + cells/ml) immediately ceased. Cells in surface scums from untreated areas drifted into treated areas, but by the time that lake wide treatment occurred all visible remnants of the bloom had disappeared. With precipitation of much of the dissolved nutrients, suspended matter, and algal cells, water clarity greatly improved. In many instances even the bottom was visible. Within five days after treatment periphyton growth accelerated. Kaufmann (1977) reported up to 10^{10} cells/m^2. Zooplankton numbers averaged about 10 per liter by December. Dissolved (.45 μm filtered) phosphorus remained low (<.01 mg/l) during the later summer-fall period and through June, 1975 when scheduled sampling ceased due to lack of funds. The

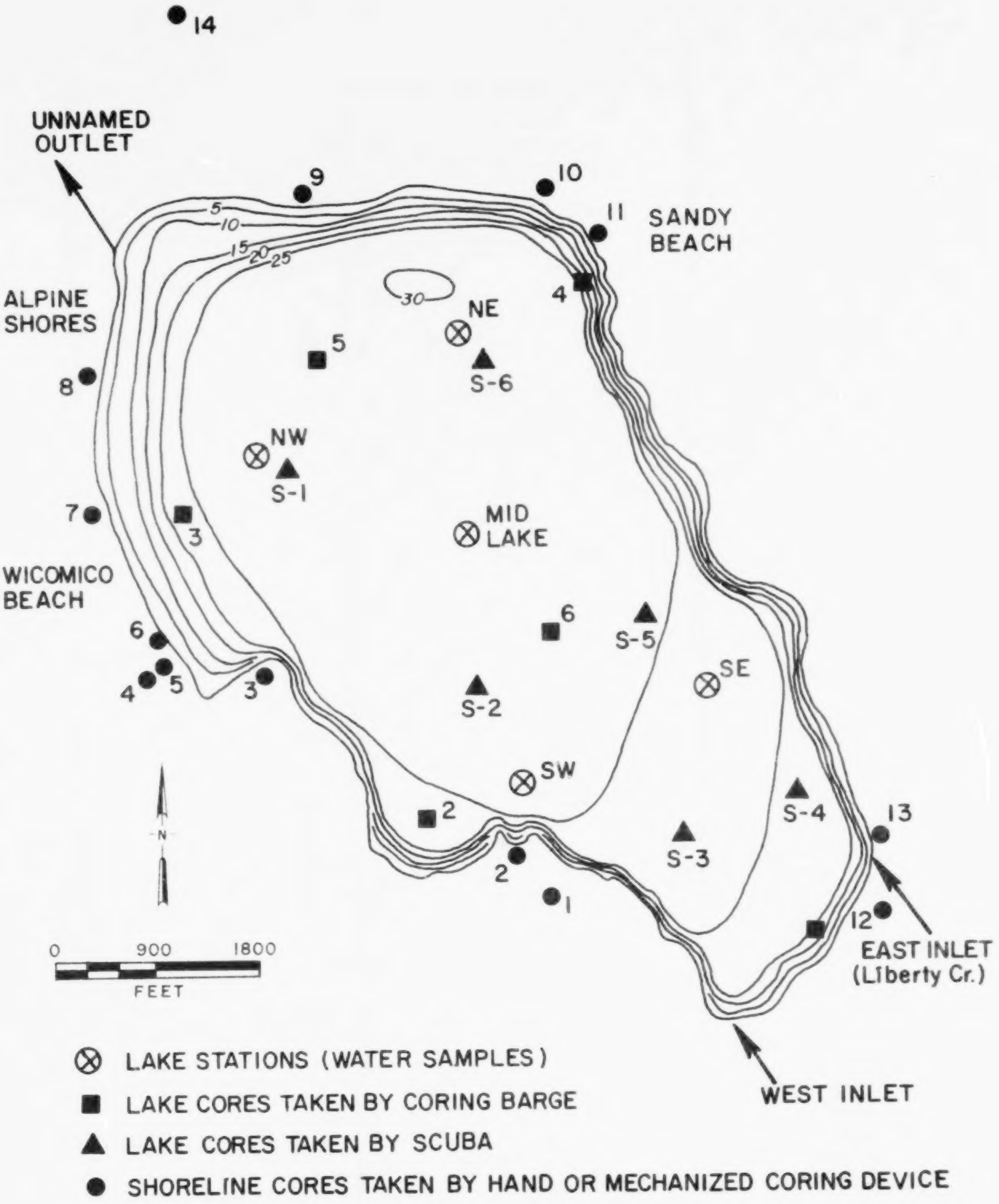
14
UNNAMED OUTLET
9
10
11
SANDY BEACH
5
10
15
20
25
30
ALPINE SHORES
NE
4
5
S-6
8
NW
S-1
7
3
MID LAKE
WICOMICO BEACH
6
S-5
6
5
4
3
SE
S-2
SW
S-4
13
2
S-3
2
1
12
N
0
900
1800
FEET
EAST INLET (Liberty Cr.)
WEST INLET
⊗ LAKE STATIONS (WATER SAMPLES)
■ LAKE CORES TAKEN BY CORING BARGE
▲ LAKE CORES TAKEN BY SCUBA
● SHORELINE CORES TAKEN BY HAND OR MECHANIZED CORING DEVICE

Figure 1. Lake water sample stations and location of core sediment samples. (Figure outline from Lakes of Washington by E. E. Wolcott.)

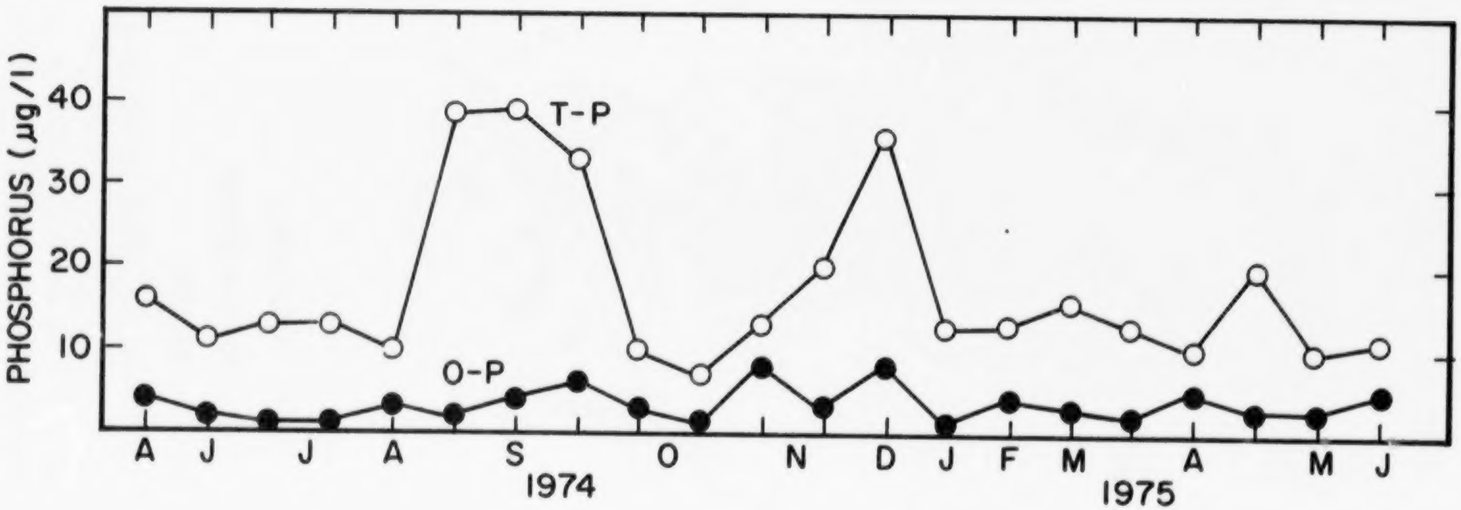

Figure 2. Ortho and total phosphorus concentration in euphotic zone, Liberty Lake, southeast station, 1974-1975.

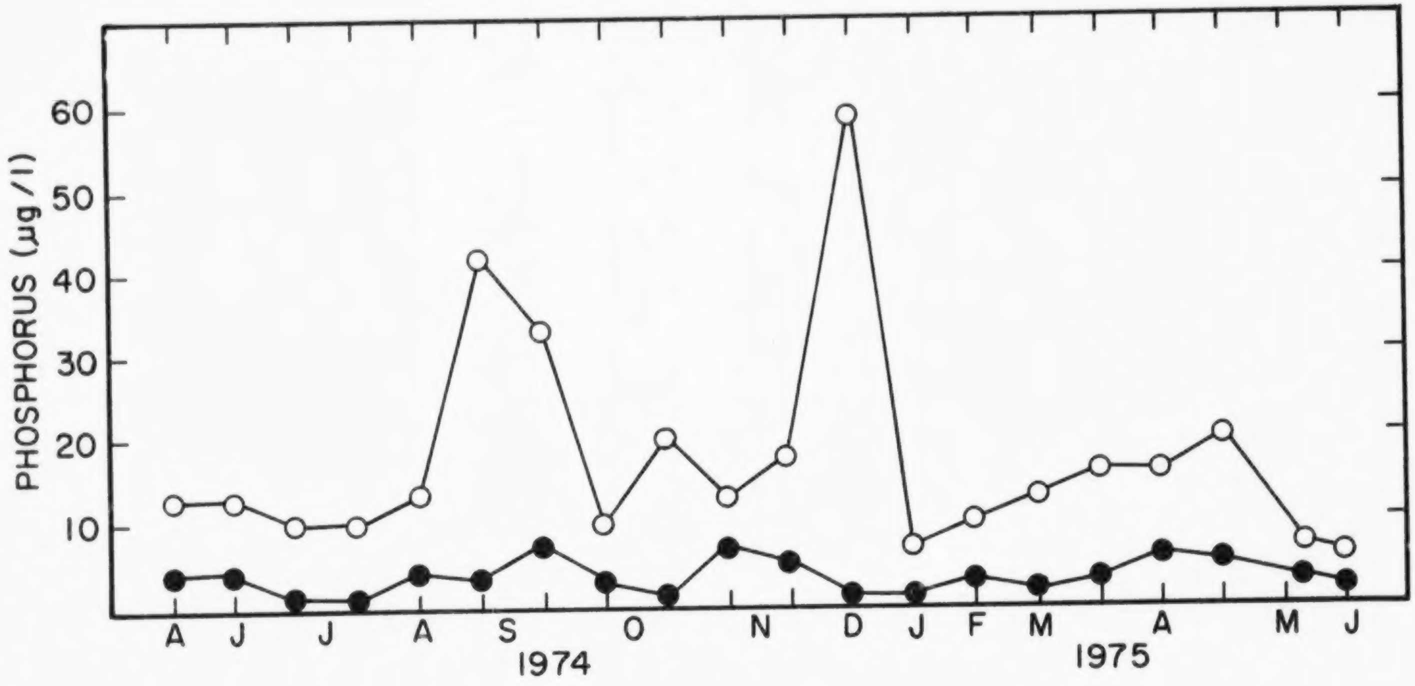

Figure 3. Ortho and total phosphorus concentration in euphotic zone, Liberty Lake, northwest station, 1974-1975.

massive blue green algae blooms that had been occurring for the past 10 years were avoided for two years following treatment. High precipitation in 1975-76, along with the macrophytes acting as nutrient pumps, helped restore the nutrient inventory, followed in turn by massive blooms of *Anabaena flos-aquae*, *Anabaena spiroides*, *Coelosphaerium Naegelianum* and *Aphanizomenon flos-aquae* in August 1977. Figures 4 and 5 show the rise in reactive phosphorus in early August at the southeast station and mid-September at the northwest station.

RESTORATION PLANS

In April 1976, the Liberty Lake community passed a bond issue for the construction of a sewer collection and treatment system for almost the entire lake. Concurrently, the lake "Ecology Committee", the sewer district and their consultants successfully proposed a lake restoration plan to the United States Environmental Protection Agency. The plan was based largely upon data generated by the studies previously described in this report. The major objective of the restoration plan was the curtailment of excessive nutrient flow to the lake (chiefly phosphorus and nitrogen) and, secondarily, the reduction of nutrient recycling within the lake. Special emphasis is being placed upon phosphorus because of the successful alum precipitation experiment of 1974.

Sewage collection and diversion is expected to be completed by early 1979. It is expected that leaching from sump holes and septic tank fields will continue for about seven years.

In-lake restoration plans include partial drawdown during a fall period, and excavation of nutrient rich sediments from the shoreline. These procedures will be followed by shallow suction dredging of about 80.9 ha (≅200 acres) of the lake bottom.

Following dredging, precipitation by aluminum sulfate treatment is proposed to: (1) remove phosphorus released from sediments, (2) reduce turbidity caused by dredging activity.

In order to reduce the level of nutrient from stream inflow, it is proposed that the stream channels be cleared of debris and deposited materials that cause excessive overflowing and flushing of the marshlands to the lake. Diversion gates would be installed to maintain water levels in the marsh. Finally, repair and reconstruction of the dike separating the marsh and lake to further reduce free movement of nutrients from the marsh to the lake. Figure 6 outlines areas in the rehabilitation plan.

ASSESSMENT OF LAKE RESTORATION PROCEDURES

PURPOSES

The proposed study will attempt to measure the effects of lake rehabilitation by observing certain biological, chemical, and physical parameters for one year prior to lake manipulation and for two years following rehabilitation.

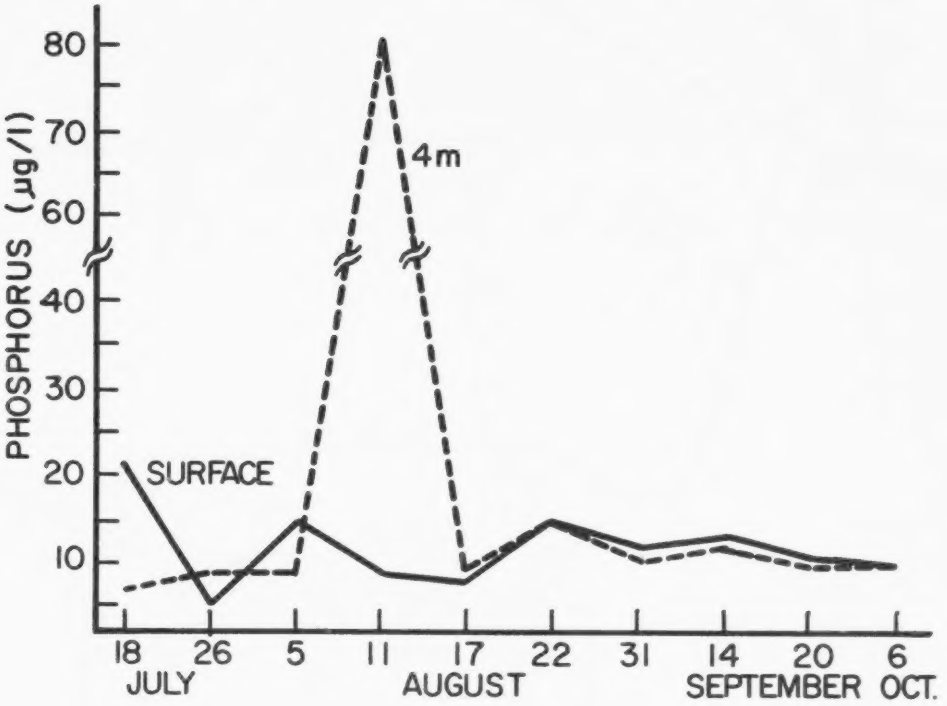

Figure 4. Total reactive phosphorus Liberty Lake, southeast station, July-October, 1977.

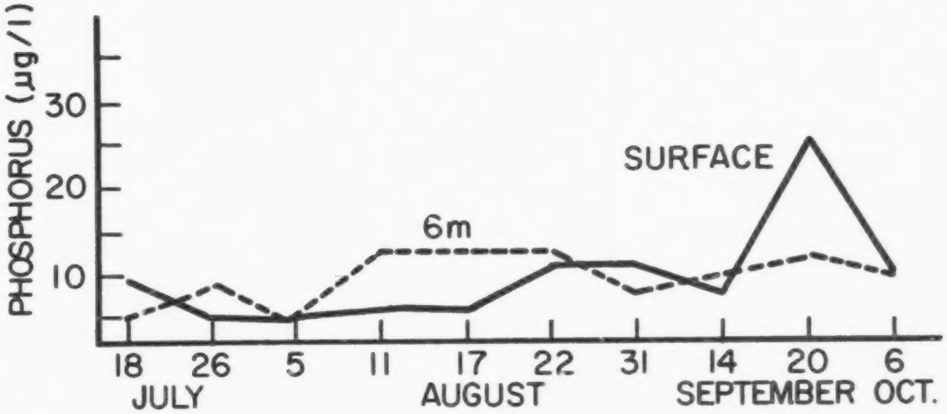

Figure 5. Total reactive phosphorus Liberty Lake, northwest station, July-October, 1977.

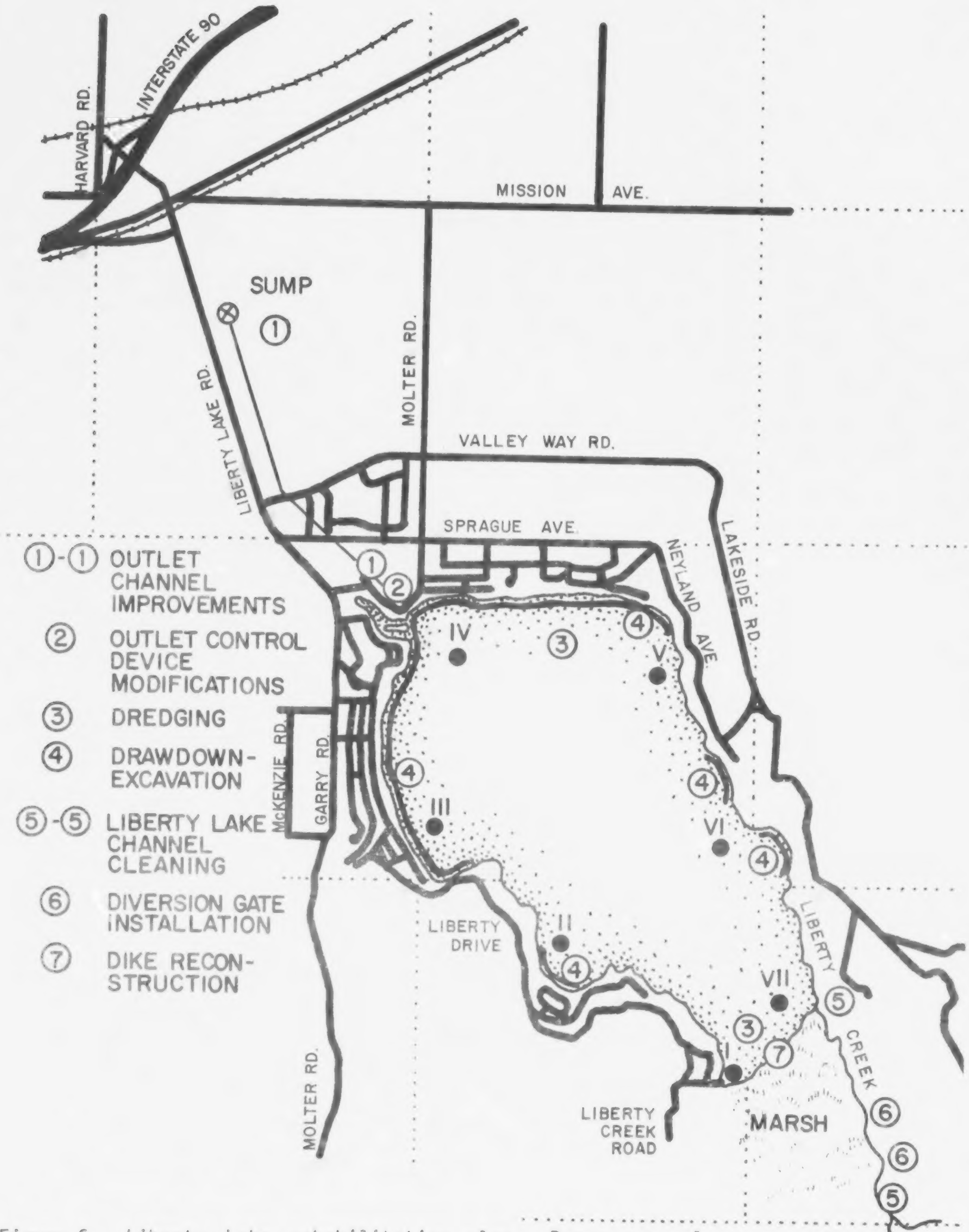
INTERSTATE 90
HARVARD RD.
MISSION AVE.
SUMP
①
LIBERTY LAKE RD.
MOLTER RD.
VALLEY WAY RD.
SPRAGUE AVE.
NEYLAND AVE.
LAKESIDE RD.
①-① OUTLET CHANNEL IMPROVEMENTS
② OUTLET CONTROL DEVICE MODIFICATIONS
③ DREDGING
④ DRAWDOWN-EXCAVATION
⑤-⑤ LIBERTY LAKE CHANNEL CLEANING
⑥ DIVERSION GATE INSTALLATION
⑦ DIKE RECONSTRUCTION
McKENZIE RD.
GARRY RD.
IV
V
VI
III
II
VII
I
LIBERTY DRIVE
LIBERTY CREEK
MOLTER RD.
LIBERTY CREEK ROAD
MARSH

Figure 6. Liberty Lake rehabilitation plan. Roman numerals represent proposed sediment core sites for assessment study.

Careful monitoring of the rehabilitation project should provide information for practical application to other eastern Washington and northern Idaho lakes as well as other lakes in the United States.

Another broad objective would be that of public education regarding the problems and cures for lakes suffering from heavy population and recreational pressures.

Tangible benefits such as reduction or elimination of algal blooms, less aquatic weed growth, improved water clarity, and general esthetic improvement of the lake would be easily recognized by the public.

SPECIFIC OBJECTIVES

1. Recalculation of prime nutrient budget (P&N) of the lake based upon measurement of nutrient inflow.
2. Attempt to quantify septic tank seepage and inflow of groundwater by use of seepage meters as outlined by Lee (1977).
3. Estimation of phytoplankton productivity and species change before, during, and after utilization of each renovative technique.
4. Determine nutrient content (N&P) of sediments to be dredged before dredging, and that of the new layer of sediments exposed after dredging--as well as the nutrient content of waters overlying these areas, before and immediately after dredging operations.
    a. Analyses of segmented core samples would be helpful in determining if dredging were useful (depth to which sediments should be removed).
    b. Cores would be of value in predicting success of this lake renovation method in terms of nutrient budget removed.
5. In areas where large beds of aquatic weeds will be removed as a result of the dredging of nutrient containing sediments, study quadrants will be established to determine regrowth rates.
6. Dike reconstruction area at the southern end of the lake will be monitored by aerial infra-red photography to observe change in weed growth patterns when seepages through breaks are eliminated.
7. Aerial infra-red photographs of Liberty Lake will also be taken periodically for comparison with those taken over the years 1968-74, when algal blooms and aquatic weed masses inundated the beaches.
8. Joint seminars or evening sessions will be conducted with the Property Owners Association, sewer district, and Kennedy Engineers for information exchange, progress reports and to preserve the spirit of cooperation which has existed to date.

METHODS

The following procedures and methods have been proposed for purposes of establishing lake characteristics, recalculating nutrient budgets, and confirming earlier baseline data. In addition, it is thought that these procedures will aid in evaluating rehabilitation techniques such as the proposed drawdown, diking, dredging, and the alum treatment following the dredging operation.

CHEMICAL AND PHYSICAL PARAMETERS

1. Three permanent inflow stations to the lake will be established, an additional intermitent urban drainage stream will be monitored. Monitoring will be accomplished by automated flow weighted composite samplers. It is planned to sample these sources weekly during run-off and summer growth (bio-reactive) periods. An outlet station will also be sampled during flow (May-June).

2. Two lake stations will be established and sampled at 2 m intervals or more frequently, if necessary, during the period of weak-moderate stratification (usually July-September). Each station will be sampled weekly during the months of May through October, and then monthly for the remainder of each year. An exception to this procedure will be made after certain rehabilitation techniques have been instituted, such as dredging, and immediately after alum treatment. At these times, intensive short term sampling for phosphorus, aluminum, sulfate, conductivity, alkalinity and pH will be undertaken. Other exceptions will be during periods of high turbulence and runoff. At such times, sampling frequency may be increased to several times per week for phosphorus components.

3. Phosphorous, because of its dominant role in controlling lake productivity, will receive special attention. Components determined would be total, total dissolved and soluble reactive phosphorus. Correspondingly, the other major nutrient, nitrogen, would be determined as nitrite, nitrate and ammonia nitrogen.

4. Routine water quality parameters measured will be:

    a. Temperature
    b. pH
    c. Dissolved Oxygen
    d. Conductivity
    e. Turbidity
    f. Total Alkalinity
    g. Sulfate
    h. Chloride
    i. Total Hardness
    j. Calcium Hardness
    k. Sodium
    l. Potassium
    m. Aluminum
    n. Iron
    o. Calcium
    p. Magnesium
    q. Silica

    (Parameters "a" through "f" would be determined weekly during May to October; parameters "g" through "m" would be determined at least monthly.

BIOLOGICAL PARAMETERS

1. Phytoplankton samples for qualitative and quantitative enumeration will be taken by continuous pump sampler at the same time and at the same lake stations that weekly chemical-physical measurements are made. One to six liter samples will be collected in the euphotic zone in accordance with methods described in EPA - Biological Field and Laboratory methods (EPA, 1973).

2. Zooplankton sampling will be carried out at a minimum of two lake stations, at the same time that weekly chemical-physical measurements are made. Collection will be made at 2.0 m intervals from surface to bottom by rapid continuous pump sampler passing waters through #10 and #20 plankton nets. One oblique tow at each station will be made by Clarke Bumpus Sampler equipped with flow meter.

3. Chlorophyll "a" samples will be taken for analysis by continuous pump sampler at each lake station and depth where water samples are collected. At least one liter at each depth will be collected and immediately treated with magnesium carbonate. Collection and analysis will be similar to that described in EPA - Biological Field and Laboratory methods.

4. Carbon 14 _in situ_ lake productivity measurements will be made on a bi or tri weekly schedule at each lake water quality station during the May through October period. Incubation will be carried out at three depths through the euphotic zone for four hours. Incubation bottles will be in triplicate at each depth. Procedures followed will be that given in APHA - Standard Methods 14th Edition (1975).

5. An estimation of the extent of aquatic weed beds will also be made by SCUBA procedures during the late summer period for an estimation of maximum standing crop. Other specialized studies will be carried out as described in the Assessment Objectives to observe the effect of the dike repair at the southern end of the lake and in the dredged areas. Steel quadrants of 1 sq m will be fabricated and located randomly in these areas. They will be harvested at selected intervals to measure biomass. The same procedures will be carried out to measure regrowth of areas exposed during drawdown.

6. Benthic invertebrates will be cataloged bi-monthly from each quadrant of the lake from April to October by Ekman grab sampler. Three to six random samples will be taken at each station. Organisms will be collected by passing sediments through a U.S. Standard #30 Sieve. Number of samples will be increased or decreased after a baseline survey, as suggested by APHA - Standard Methods (1975).

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ECONOMIC IMPACT OF LAKE RESTORATION
LIBERTY LAKE, WASHINGTON

by

K. C. Gibbs and L. E. Queirolo*

INTRODUCTION

In general, as a lake ages it undergoes changes and a natural maturation process takes place. Limnological research being done is directed toward finding technically feasible methods of improving water quality. At some point, however, enhancement of a lake must be conceptualized in social rather than physical terms. Methods of improving and maintaining water quality, when established, will have to be acceptable to or desirable for the people and be worth doing in order to be implemented. In other words, protecting or improving water quality, fish spawning grounds, or waterfowl habitat are not in themselves the ends of social policy.

Whenever decisions are made on a broad policy issue, such as restoring water quality in a lake, some individuals or groups will benefit and some will incur detrimental impacts. Given the scarcity of available resources for water quality improvement, it is imperative that they be devoted to projects where the payoff, in terms of benefits, is greatest. The economic evaluation of water pollution control is often difficult, especially if, as in the case of Liberty Lake, Washington, many of the benefits are in the nature of "extra market goods," such as outdoor recreation.

Liberty Lake is situated about 13 miles east of Spokane, Washington. The lake is primarily a recreational lake, 781 acres in size and receives runoff from a 13.3 square mile watershed. It is the purpose of this proposed study to estimate the significant economic impacts (not necessarily recorded in a market, but in terms of what a person or group would be willing to give up to have higher water quality) on recreationists, and on adjacent and nearby landowners at Liberty Lake.

As a result of a delay in project funding, research on the economic impact of lake restoration at Liberty Lake will begin March 15, 1978. A literature search and theoretical model formulation will then begin. Details of the project are given below.

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RESEARCH PROJECT
ECONOMIC IMPACT OF LAKE RESTORATION

RESEARCH CONTEXT

In many lakes, accelerated growth of biological organisms has resulted in levels of water quality restricting or hampering the use of this water resource for certain recreational pursuits. Research is being carried out by the Environmental Protection Agency and others on the physical and biological aspects of this enrichment process.

In general, as a lake ages it undergoes changes and a natural maturation process takes place. Precipitation and natural drainage contribute nutrients which support and facilitate the growth of vegetation within a lake. The extensive activities of man, however, can increase the amounts of nutrients deposited in a lake in several ways: by a more intensive use of the agricultural land; by urbanization; and by the discharges of industrial wastes, and waste treatment plant effluents. The process of enrichment of waters with nutrients that occurs naturally is often accelerated by man's activities. The resulting quality of the water may thus change significantly and often at a relatively rapid pace. Some recreational activities may be discouraged (such as swimming) while others (waterfowl hunting) may be facilitated.

The limnological research being done is directed toward finding technically feasible methods of improving water quality. At some point, however, enhancement of a lake must be conceptualized in social rather than physical terms. Methods of improving and maintaining water quality, when established, will have to be acceptable to, or desirable for, the people and be worth doing in order to be implemented. In other words, protecting or improving water quality, fish spawning grounds, or waterfowl habitat are not in themselves the ends of social policy.

OBJECTIVES

Whenever decisions are made on a broad policy issue, such as restoring water quality in a lake, some individuals or groups will benefit and some will incur detrimental impacts. It is the purpose of this proposed study to estimate the significant economic impacts (not necessarily recorded in a market, but in terms of what a person or group would be willing to give up to have higher water quality) on recreationists, and on adjacent and nearby landowners at Liberty Lake, Washington. More specifically, the objectives of this proposed study are to examine Liberty Lake and:

1. Refine current methodologies and estimate the economic value of lake restoration to recreationists engaged in various water oriented activities.

2. Estimate the economic impact of lake restoration to adjacent property owners.

3. Identify and evaluate costs of lake restoration.

It is very important that any research done on the economic impact of lake restoration coordinate with the other two significant aspects—namely the limnological and other social impacts. This will be a primary goal in this study, to attempt to keep communications open with personnel performing physical work on Liberty Lake (see Funk, 1975; and Kennedy, 1977) and with the EPA funded project to estimate the sociological impact of lake restoration on Liberty Lake (Honey and Hogg, 1977). This represents a unique opportunity that the limnological, economic and other social aspects on a single lake could be coordinated where each discipline gains from the others and a more realistic product is the outcome.

Given the scarcity of available resources for water quality improvement, it is imperative that they be devoted to projects where the payoff, in terms of benefits, is greatest. The economic evaluation of water pollution control is often difficult, especially if, as in the case of Liberty Lake the benefits are in the nature of "extra market goods," such as outdoor recreation.

This proposed research is directed toward evaluating the economic benefits resulting from increased utilization of water resources for outdoor recreation. This is important for at least two reasons: First, it provides a guideline for decision-makers concerned with the allocation of public funds for water quality improvement, in the case of Liberty Lake. Second, it is anticipated that the methodologies developed in this study will be useful in the evaluation of recreational benefits resulting from water quality improvements in other cases. In regard to the latter point, it should be noted that some recent developments in economic analysis have provided for the estimation of the demand for outdoor recreation. The theoretical models, however, need to be developed further to permit an application to a more diversified range of problems.

STUDY AREA

Liberty Lake is situated about 13 miles east of Spokane, Washington. Liberty Lake is primarily a recreational lake, 781 acres in size, and receives runoff from a 13.3 square mile watershed. The lake occupies a shallow basin with a maximum water depth of 30 feet and a mean depth of 23 feet. It is fed by a perennial stream entering through a marsh at the upper end of the lake. The south end of Liberty Lake has a gradually sloping bottom and supports significant amounts of aquatic weeds. During the summer months, these weeds reach nearly to the surface as far as one-third of a mile from the south shore.

Liberty Lake is classified as a shallow, soft-water, meso-eutrophic lake (Funk, et al., 1975). It is these characteristics which play an important role in impacting the current use of the lake. With increased growth of Spokane in the last several years came increased pressure on the Liberty Lake community. This community was once a summer resort and rural-agricultural community. It has since absorbed rapid growth from the Spokane Valley.

Land adjacent to Liberty Lake is used primarily for residential purposes. In addition, a wildlife refuge and outdoor recreation area is maintained to the south. The trend is toward suburban-type residential and recreational development. This area is being looked at as a desirable location for further development. Several resorts are located within the proximity of Liberty Lake and some people live here and commute to their jobs in Spokane. Other establishments in the community include several taverns, a grass seed growing business, and a grocery store.

Recreational activities on Liberty Lake include trout fishing, swimming, camping, hiking, picnicking, and water skiing (thought not to be observed on other lakes in the close vicinity of Spokane. See Kennedy, 1977). Swimming and fishing are the two most important activities on the lake.

This lake was chosen for study primarily because both limnological and sociological aspects are currently being studied on Liberty Lake. Much relevant physical data were already accumulated. This is helpful in that economic impacts can be related to known conditions, and variables in models can be more realistically specified. In addition, close coordination with the sociological aspect can be maintained.

PROCEDURE

While a more thorough literature search and review is required to fully utilize the state of the arts and to determine what methodological and modifications might be required, preliminary findings suggest the following approach.

OBJECTIVE 1

Background

In order to estimate the benefits to recreationists of lake restoration, a demand relationship is needed. This relationship is composed of the quality demanded as a function of price, income, price of substitutes and tastes and preferences. The difficult problem when dealing with a commodity, such as outdoor recreation, that is publicly provided or otherwise consists mostly of common-property resources, is the lack of a price (or at least a negligible fee with a significant variation). To estimate the demand for a non-market good or service, the consumers' reaction to price increases is simulated either by evidence gathered from direct questioning or by observing their reactions in already existing and related markets.

Procedures have been developed to estimate demand curves using both general procedures. Direct questioning methods have been used with some success and seem appropriate in certain instances where no expenditures can be observed in any subsidiary markets. (Knetsch and Davis, 1966) (Pearse Bowden, 1970 and 1971). In these cases, asking recreationists what they would be willing to pay, rather than do without the activity, must be done in a careful manner not to get hypothetical answers. Biases, tainting the accuracy of the estimates, must be guarded against. This procedure will be evaluated for use in this study.

The other general category of procedures to estimate demand for recreation, the indirect observation of expenditures in related markets, deserves further attention. These types of methodologies involve the use of some surrogate, or proxy for price. More precisely, the recreationist's willingness to pay is based on observations of costs actually incurred to recreate at a facility. Hotelling (1947) is credited with the original idea of using travel cost (the cost of overcoming distance between the facility and a series of more or less concentric distance zones between the facility) as a proxy for the price of a visit to the facility, although Clawson (1959) provided the first application of Hotelling's idea. Later variations and refinements of the so-called "travel cost approach" include Clawson and Knetsch (1966), Brown et al. (1964), Burt and Brewer (1971), and Pearse (1968). Most advocates of this method incorporate travel costs as well as on-site costs as the price variable. In addition, income, distance, time, and other socio-economic variables are included in the analysis. Two relationships are estimated: one representing the total recreational experience (including travel, anticipation, recollection, and actual time on-site): and the second, derived from the first, to estimate the responsiveness in the quantity consumed of changes in a user fee.

Some shortcomings of the traditional travel costs approach have been raised in the literature (see Edwards, et al., 1976; Gibbs, 1969; and Jennings, 1975), and as a result, variations of the indirect method have been developed. In these, alterations have been made with respect to the price variable. Total trip costs are divided into travel costs (all costs incurred while enroute to and from the facility) and daily on-site costs. These two components are then expressed as separate explanatory variables, with on-site cost the choice of the facility proxy.

Whatever their differences, a major common feature of all the indirect approaches is the assumption that the price of using a recreational facility can be reasonably represented by the costs of certain goods and services that are purchased in conjunction with facility use.

Model

This study would first analyze the types of recreation occurring on Liberty Lake, characteristics of the recreationists (travel distance, etc.), and related services provided. Then, coupled with a thorough literature review, devise a theoretical framework to estimate recreation demand in this area. Total recreational usage can be defined as the product of the number of days a recreationist uses a recreational site per visit and the number of visits to a recreational site. If both the length of stay and the number of visits are variable and reflective of water quality, then two relationships should be estimated with each as independent variables. The theoretical model postulated herein will have the following general form with possible modification based on further study:

$$DV_i = f\ (C,\ T,\ S,\ WQ,\ SE)$$

$$V_i = f\ (C,\ T,\ S,\ WQ,\ SE)$$

$$\text{Total Usage} = DV \cdot V$$

Where Dv_i is the number of days per visit the recreationist uses the lake

for activity i, and V_i is the number of visits a recreationist makes per year to participate in activity i. The unit of measure is the recreation group since this is the decision-making unit rather than an individual or family (more and more non-family groups are enjoying recreational activities). The primary water-related activities are swimming, fishing, boating, and lake related camping. These will be accounted for separately since a change in water quality would impact each of these activities differently.

> The following explanatory variables are thought to be the ones to most significantly explain days per visit and number of visits. C is the daily on-site costs incurred by the group participating in a particular activity. These are the costs to which a recreator reacts in deciding how many days to recreate. T is the group's travel cost for each visit. This cost is fixed with respect to the number of days at the site per visit but variable when considering the number of visits to make.
>
> S is used to represent a variable to account for substitution among activities and lakes in the study region. This will be examined and specified in more detail during the study period.
>
> The various degrees of water quality are to be represented by WQ. These will include those outcomes of water quality improvement that affect each activity. For example, it is not the presence or absence of nitrogen that induces a swimmer to participate elsewhere, but the biological effect of N; e.g., blue-green algal blooms. These variables will be defined in association with EPA personnel, limnologists, and recreationists in the area. Recreationists will be asked how their use, in terms of the number of trips, length of stay, and resulting activities, will vary as the water quality improves.
>
> SE refers to a set of socio-economic variables found to significantly influence recreation use in the area. These may include items such as income, age, size of group, destination visitor, equipment utilized, amount of recreation-related time per year, etc. A further study of the area, users of the area, and past studies will lead to the selection of the variables used here.

A sample of recreationists will be drawn at public access points surrounding Liberty Lake. Measures of the variables in the model will be obtained via a personal interview. Enough recreationists will be contacted to ensure a statistically sound demand estimation. In addition, water quality variables will be obtained from secondary sources.

After estimating the demand model, estimates of value will be made utilizing consumer surplus. The local expenditures will also be tabulated to recognize the impact of recreationists on the local communities.

OBJECTIVE 2

In addition to the impacts of an improvement in water quality on those recreationists utilizing the lake's resources via public access, another segment of the population gains value through the appreciation of private property values. One of the most important sources of land value increases around a body of water is the value as a recreational or aesthetic resource. The increment to the value of property attributed to the lake is an expression of the benefits derived from the water. Higher land values adjacent to lakes are hypothesized to represent a capitalization of a portion of these benefits. These land values will be sensitive to the quality of the water in its proximity. This portion of the study will utilize a methodology to estimate the increase in property values attributed to an improvement in the quality of the water adjacent to or near the property.

Several past studies have concerned themselves with the identification and relative significance of factors which affect the values of residential property. Jack L. Knetsch (1964) reported that land bordering surface water does have incremental value attributed to the presence of a reservoir or artificial lake. He compared land with water frontage to similar land without water frontage to observe the difference in the per acre sales price of individual parcels.

David and Lord (1969) reported that land bordering surface water does have incremental value attributable to the presence of a reservoir. Their study was concerned with determining the extent to which certain characteristics influence the demand for recreational land on artificial lakes. Improvements to the property were included in the value of the tracts.

Research by Schutjer and Hallbert (1968) indicates that capitalization of recreational facilities of water based state parks into local land values has occurred. Taking observations on transfers before and after the development of a reservoir, they observed the influence of water-recreation availability on land prices in the nearby area. They used multiple regression analysis with 15 independent variables on the sales observations of the same tracts of land before and after the development of the park.

Connor, et al. (1973) used two methods of estimating the value of the presence of water frontage to typical residential property in the Kissimmee River Basin, Florida. The first used multiple regression to analyze the effect of several independent variables, including lake frontage, on vacant residential lot sales. The second estimated the value attributed to the presence of water frontage from owners' estimates of the value of their property (with houses) with and without water frontage.

The first step in accomplishing this objective is to make a thorough search of the literature to gain insight into models that have been used (and their success) to estimate the value of improvements in water quality and/or the presence of water on property values. Most of the land adjacent and near Liberty Lake, especially on the northern end, is used for residential purposes. This ownership is where the primary impact on property values will occur. Thus, an estimate of the increase in residential property values will be estimated. Benefits to resorts, and those charging fees to the public for recreational purposes, will be reflected in their receipts.

The model to estimate the increase in property values proposed at this time, but subject to refinement as more information is obtained, is generally as follows:

$$Y = f\ (Yr,\ Ls,\ WF,\ P,\ T,\ WQ)$$

A sample of residences will be drawn from those immediately adjoining Liberty Lake and those in the proximity of the lake. Actual sales of property, with and without structures, will be analyzed in the basin. In addition, personal interviews will be conducted with those land owners in the sample to derive information of their perceptions of the impact of a change in water quality. The variables in the model are defined as:

> Y is the sales price of the property expressed either as a total price or on a per acre basis depending on the type of property considered. If sales records are not adequate, this variable will be estimated utilizing appropriate questions posed to owners of property.
>
> Yr is the year either of the sale or date of evaluation of this piece of property. This is expected to have a positive relationship to sales price since sales prices have increased with inflation and the expanding demand for this type of property.
>
> Ls, the size of the lot, is measured in the number of acres. It is hypothesized to have a positive relationship with total sales price, but negative with respect to the price per acre.
>
> WF is defined as the distance of the property to the lake. This is importance since the mere presence of water has a significant impact on the sales price. But, a change in water quality has an impact on land near but not adjacent to the lake. This impact will be different, it is hypothesized, depending on the relationship of the property to the lake.
>
> P represents the proximity of the land in the area. This could include variables such as distance to paved roads, access to the property, utilities available, and so forth. The specific variables will be identified upon a more thorough examination of the area.
>
> T refers to the types of structures on the property. This would have an impact on the property value that needs to be accounted for even if vacant lots were analyzed.
>
> WQ is the water quality variable. This variable, as in the case of the model utilized on visiting recreationists, will be defined on the basis of the outcomes to which participants are responsive. The exact formulation of this measure will be identified after close association with EPA personnel and local individuals.

After data collection, the model will be estimated using multiple linear regression analysis. From the estimated relationship, the change in the property value associated with changes in water quality (WQ) can be estimated. This influence can also be isolated on different lot sizes, for different types of developments, types of structures, etc. That is, by manipulating the values of the other independent variables in the model, the influence of water quality can be estimated for different situations that may occur in other areas. Thus, the model will serve as an attempt to draw some conclusions of the impact under various circumstances, even though it must be kept in mind that the data are from one specific area. Applicability to other areas does exist, however.

Other impacts of water quality improvements on adjacent communities, primarily of an indirect nature due to increased expenditures in the area, may be important in some areas. However, it is believed that, in the case of Liberty Lake and the nearby region, a change in water quality will not appreciably increase expenditures in the local area. The lakes in this area are not nationally or even regionally known. They are utilized primarily by local or nearby residents. Upon lake restoration activities taking place, few additional persons from outside the area will be attracted to the lakes. Thus, an increase in economic activity will likely not be significant.

OBJECTIVE 3

In addition to recognizing benefits received from restoring a lake, costs must also be identified. These come in two main categories: the initial cost of improving the lake, and the alterations, either structurally or non-structurally, required to maintain the increased quality.

Initial costs can be estimated based on limnological research being conducted in the study area and the degree to which a lake is to be improved. These procedures will be identified in close consultation with EPA personnel, other limnologists, and other individuals. It is anticipated that lake restoration would consist of a "vacuuming" of the lake to remove the vast amount of aquatic weeds and nutrients accumulated and then in addition a treatment of alum. Costs of this treatment will be calculated based on work under way by William Funk, limnologist at Washington State University.

In addition to treatment costs or removing excess nutrients from the lake other changes are needed to slow the nutrient input into the lake. This can be accomplished by changing the waste disposal activities of the adjacent residents. A switch from septic tanks to a central waste disposal plant is required. The cost of this will be estimated and capitalized over its expected life.

Other ways of reducing the flow of nutrients into the lake, such as less lawn and garden fertilization, will be attempted via an educational system to induce residents to want to change their patterns.

After results are obtained from all three objectives, estimates of economic benefits and costs of lake restoration will be available for Liberty Lake, Washington. The economic feasibility of cleaning the lake can then be assessed. The other social impacts must also be evaluated and integrated into the decision to clean a lake or not. As a result of this proposed study, procedures will be available to estimate the benefits and costs of lake restoration at other locations.

SUMMARY

A final report will present the data in tabular form, summarize the most important findings, and make recommendations as to its appropriateness and application for future use.

Probable Duration

24 months: February 15, 1978 - February 15, 1980

University Units Involved

Forest Research Laboratory, Resource Recreation Management Department - research staff, equipment and services.

Agricultural Experiment Station, Agricultural and Resource Economics Department.

Cooperation

U.S. Environmental Protection Agency

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SOCIAL IMPACTS OF LAKE RESTORATION, LIBERTY LAKE WASHINGTON: A STATUS REPORT

by

T. C. Hogg and W. D. Honey*

INTRODUCTION

Funding notification for the social impact study of Liberty Lake was received from EPA in September of 1977. The initial phase of the research was (and still is) a review of pertinent literature and the beginning of a compilation of resources on the historical and cultural background of the research setting (Spokane and Spokane County, WA). Preliminary field observations, including contact with a number of people with special knowledge (key informants) of the project and its social parameters, started in mid-February and will continue through May 15, 1978. We, therefore, are just underway with the effort.

The major objectives of our Liberty Lake studies are to (1) further identify and describe the range and types of social impacts that are associated with lake restoration, (2) analyze the function and significances of all identifiable social impacts, including those associated with planning, lake treatment and restored lake phases and, (3) explain the process of cultural-environmental interplay which operates in the impacted setting and, (4) refine our methodology for later more precise and effective use.

The methodology employed in this social impact assessment is what commonly is referred to as cultural ecology. It entails the holistic description and analysis of relationships between human cultural systems, including values, organization and technology, and features of natural environment. Projection of future circumstances requires explanation of systems through time, i.e., a processal perspective. It is essential, therefore, to consider cultural systems in historical or evolutionary perspective (Buckley, 1968: 491). It is for this reason that historical data are so germaine to assessment. In this report we offer an overview of the phases of cultural development that preceded Liberty Lake's restoration. While they no longer operate as dominant systems, they nevertheless still influence contemporary cultural-environmental relationships. They offer an intellectual context for examining Liberty Lake in the present, and projecting social impacts of restoration into the future. Indeed, if a present social profile is instrumental to making reasonable projections of future circumstance, then the past at least provides intelligibility for the present. The implications prehistory has for the present might not be obvious, but they nevertheless raise questions for the present since they constitute alternative models of lake usage by human beings. The importance of this historical orientation will be given fuller explanation in the final part of our presentation on Liberty Lake Social Impact Methodology.

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HISTORICAL BACKGROUND

The history of human involvement in the Liberty Lake Region involves adaptive cultural systems and environmental interplay. They are (1) hunting and gathering systems represented by American Indians, (2) agrarian systems in the early Euro-American period, and (3) the industrial-urban system that first emerged in the late 1800's and early 1900's and persists to the present.

HUNTING AND GATHERING

The Coeur d'Alene and Spokane Indians were two contiguous groups indigenous to this area. Each represented a unique and quite different human adaptation. The Spokanes were a riverine adapted people; the Coeur d'Alene were oriented to lakes and their exploitation. The Coeur d'Alenes pertain directly to Liberty Lake.

Occupying the majority of present day Idaho and a portion of eastern Washington above Spokane Falls, the Coeur d'Alene illustrate a systematic human exploitation of a lacustrine province. Their technology reflected this orientation. Quite early they had developed a rod and reel apparatus (fishing pole) that was used not only for fish but also to snare ducks and geese (Teit, 1904). Lakes were also used to trap land animals such as deer. The most widely used canoe was the variety referred to as "Sturgeon-Nose." It was more adaptable to lake use in that it could withstand rough waters (Turney-High, 1941). Occasionally Coeur d'Alene used Tule reed rafts, but these were used primarily for individual hunting and fishing rather than in group or communal subsistence quests.

The only documented use of Liberty Lake by Indians is noted by Vernon Ray in a collection of testimonies obtained from aboriginal informants and reported in 1936. Ray's informants mention a Coeur d'Alene village site comprised of some thirty families at the south end of Liberty Lake near the marsh (Ray, 1936: 132). No time reference is given, however, for its occupation, but presumably it was occupied until the mid-19th century.

FUR HUNTERS AND TRADERS

The coming of white furriers and early traders started a transition period from the former hunting and gathering culture to an agrarian system in the Northwest. The initial white penetration of the Northwest and the Spokane area occurred in the 1790's with the explorations of Alexander Mackenzie of the Northwest Company. MacKenzie was instrumental in charting most of the Frazer River drainage to the north of the Spokane Valley.

Fur trade with local aboriginal groups actually did not occur until 1810 when the Northwest Company, under the direction of David Thompson, established the "Spokane House" near the confluence of the Spokane and Little Spokane Rivers (Tyrell, 1916). Earlier in 1805, the Americans, through the explorations of Lewis and Clark, and the later Pacific Fur Company, had established a stronghold in the Spokane Region, but trade had not developed. In 1813, however, American interests withdrew and by 1821 the Hudson's Bay Company emerged as the sole monopoly which was to dominate trade with Indians in the Northwest (Rich, 1950).

Canadian, American and British furriers represented an entirely new element of population and culture in the region. Their effects were profound. In addition to altering the aboriginal lifestyle by introducing a dependency for trade wares, they also succeeded in providing a stimulus to attract more Euro-American settlers and explorers. Individuals such as David Thompson succeeded at early dates in mapping and charting not only the Spokane drainage systems, but also that of the Columbia. Cumulatively, the furriers were successful in introducing market oriented exploitive behaviors among the aboriginal populations, who became unwitting front line agents of a new culture. The fur trade continued until the 1870's, but by the early 1830's it brought the introduction of a new cultural system whose advance guard came in the form of white missionaries.

AGRARIAN CULTURE

The late 1830's marked the presence of Catholic and Protestant Missionary involvements in the Spokane area (Drury, 1976:82). The success or failure of these religious endeavors is for the most part relatively unimportant for organization of people or resource exploitation. What is important is that they brought additional publicity for settlement by advertising and identifying the attractiveness and availability of abundant resources in the Northwest. Their initial concerns were for the "souls" of the heathen, but passive plateau Indians, but the societies they represented were anxious to establish white communities in the Northwest. Settlement in the Spokane region was slow because the Willamette Valley was the chief attraction to new settlers.

The year 1850 brought a culmination of previous Euro-American settlement efforts. The Donation Land Act, designed to open up Oregon to white settlement, had an enormous effect upon the still relatively isolated western United States, including the Spokane region. It served to legitimize existing land claims in some areas, but its principal impact was to stimulate settlement throughout the Northwest (Robbins, 1974). The Act served as an impetus in establishing a strong agrarian base in Washington as well.

Agricultural settlement was the primary result of demand for land and national territory. More isolated areas such as the Spokane Valley became dependent upon the development of line of transport for settlement. In 1858 the Mullen Road was built from the Columbia to the Missouri and effectively centralized trade between the two river systems (Elliot, 1923:207). It received heavy use from freight wagons, stages, miners, and settlers from the Missouri to Walla Walla, Washington.

Another stimulus to permanent settlement in the Spokane region was mining activity in the Northwest after the 1850's. It succeeded in attracting individuals from California, Nevada and other regions and necessitated a strong resource support base, one that included agriculture and lumber as well as transportation facilities and networks. Portland was a redistribution center and the Spokane River Valley initially participated only in a peripheral manner. The city of Spokane nevertheless profited from the contiguous mining operations of Central Washington, Idaho, Montana, and to some extent, Oregon from the 1860's until, actually, the present day (Pomeroy, 1965:50). This profit came from its function as a supply dispersing point to mines and by providing such support as lumbering and smelting operations.

The new white resource orientations put substantial pressures upon the previously eroding aboriginal systems. Political policies and national goals foreclosed on Indian lands. New foci of human activity emerged with the agrarian system. Farming, hand lumber operations and minerals were important to the continuing persistence and survival of the new system. Liberty Lake *per se* did not receive commercial white attention or settlement until the 1870's when a retired Hudson's Bay Company trapper took residence near the lake. He continued trapping operations while engaged in subsistence farming (Meany, 1937). By the mid-1870's other people came to Liberty Lake and engaged in small-scale farming/ ranching operations near the lake's margins. During the 1880's it is noted that some logging and road building activities also occurred near the lake (Kennedy 1977:23).

INDUSTRIAL-URBAN

The late 19th and early 20th centuries marked the onset of the industrial-urban cultural system. It is represented by sophisticated means for harnessing energy and large concentrations of people. The railroad, water power/irrigation, and industrialized mining were especially important to the Spokane Valley. The industrial-urban systems required a reorientation of people's value and attitudes toward resources and settlement patterns.

The growth of Spokane and the adjacent area are attributed to the natural resource potentials. It possesses water for power and irrigation, lumber, good soils, a mild climate and strategic location (Meany, 1946). The discovery of minerals in the Coeur d'Alene Mountains had a dramatic effect upon Spokane's industrial development and population growth. By 1889, Spokane had nearly 25,000 residents, a 2500% growth in some 5 years (Fargo, 1950). In 1890, a dam was constructed on the Spokane River which provided hydroelectric power to the populus as well as to associated industrial developments.

The railroad perhaps was the greatest single stimulus for the area in growth and urban development. The 1880's marked the establishment of two transcontinental railroads and several interregional lines for Spokane (Gilman, 1923). It not only supplied mining operations, but also brought more permanent settlers, and increased trading potential.

Attention of the industrial-urban system to Liberty Lake emerged on or about 1894 with the first commercially promoted recreational activities. By 1899 irrigation development emerged from the lake to provide water for agriculture in the eastern Spokane area. An electric railroad was established in 1905 in order to transport individuals to the lakeside for recreational pursuits. During this same time span, ranching and farming continued to develop near the lake area (Kalez, 1973).

It was not until World War I that attention was directed toward other resources of the Spokane region. This was due in part to the emergence of an improved highway system and the automobile. Eventually it was necessary to abandon the railroad due to infrequency of use. In addition, small scale housing developments appeared on the western portion of the lake shore.

At present we have not obtained sufficient documented material to discuss the period from the 1930's through the 1960's. We will, therefore, discuss some of the social and political events leading up to the restoration program for Liberty Lake.

SOCIAL AND POLITICAL EVENTS LEADING TO RESTORATION OF LIBERTY LAKE

This immediate chronology was gleaned from several sources: 1) from newspaper articles, 2) from brief interviews with key informants involved in, or who have promoted, the Liberty Lake restoration program, and 3) from unpublished notes of people involved.

By the early 1950's, Liberty Lake residents became aware of serious algae blooms in the lake. As a result of more severe late blooms, the lake residents in 1968 established a group called the Liberty Lake Ecology Committee. Upon their formation, they sought assistance to determine the extent of natural and cultural aging of the lake. Limnologists from Washington State University prepared a report on several recreational lakes in western Washington, and from this report, an issue emerged as to whether lake aging actually meant that there was a bonafide water pollution problem at Liberty Lake. Individuals as well as agencies took opposing sides in this issue, but one thing was clear--there was a real use decline on many of these recreational lakes.

In the early 1970's, the first systematic study of Liberty Lake was conducted by Washington State University. From the results, the Ecology Committee concluded it was necessary to implement sewering. The committee reorganized itself into the Liberty Lake Sewer District. It immediately sought to enlist public support and formulated the Annual Ecology Day which concerned itself with the cleanup of debris on the shoreline. In 1974 a sewer plan was prepared and action was taken for a bond election. Countergroup activities formed in opposition of the bond election. The "Committee of Concerned Liberty Lake Tax Payers" effectively counteracted. Action was not successful as only the sewer plan and the revenue assessment passed. The general obligation bond failed. This was an important setback, however, since the obligation bond was needed to implement the sewer system.

In preparation for a March 1975 election, a reconsideration of the general obligation bond occurred, but countergroup action was again successful in defeating it. More careful planning was given to the issue for the November 1975 election. The bond amount was reduced, the committee solicited endorsements from EPA and the County Commissioners. Countergroups action was again strong, and the bond was once again defeated.

In March 1976, however, they were able to secure federal and state funds. A Corps of Engineers report stated that the Spokane Aquifer was being polluted from septic tanks in the area provided the necessary impetus to obtain passage of the bond. In July of 1976 the Washington State Department of Ecology held hearings to hear requests for lake restoration projects. Although some opposition emerged, favor and support emerged for Liberty Lake. In February of 1977, EPA approval was granted to Liberty Lake and the project emerged as the first Clean Lakes Program in the western United States. Early proponents of this program envision it as becoming a model for urban development in a rural area.

HISTORICAL SUMMARY

The evolution of cultural systems in the Spokane River Valley and Liberty Lake area must be viewed in a progressive and processal sense. The technological base shapes attitudes toward the ecological or environmental system. History reveals that the more efficiently energy is harnessed the more elaborate and exploitive the systems become. Values, attitudes, and orientations towards the region's resources dramatically shift from population to population.

Thus, we come to the point of our entry into inquiry on the social impacts of Liberty Lake's restoration. Historical data suggests industrial-urban growth to be a major factor (or set of factors) in accelerating eutrophication of the lake. Whereas conditions undoubtedly created water quality problems at earlier points in time, these either were naturally alleviated or ignored until sufficient density of settlement, demands for unrealized property value potential, or recreational usage created recognition of water quality problems, and demanded organization and action to correct them.

Our research effort enters the scene at a time when lake treatment is impending and restoration is incipient. It is precisely at this point that it becomes important to describe the nature of the research, together with its anticipated results. The next phase of this paper will offer a description of our methodology.

LIBERTY LAKE METHODOLOGY

GENERAL

Assessment of the social impacts of public works projects represents a new concern and activity. Legislation of the 1960's and 1970's has required public agencies to adopt research programs to evaluate the "overall" effects of their developmental programs, including the social parameters of such development. Shortly after the passage of the National Environmental Protection Act in 1971, different federal agencies developed separate guidelines for such research and social impact assessment lacked direction. Most important as a correction to this problem was the Water Resource Council's Establishment of the Principles and Standards (volume 38, no. 174 of the Federal Register, 1973). This document attempted to unify objectives of all federal agencies, especially with regard to assessing project impacts on the quality of life and social well-being of a related population.

As this workshop attests, the social impact assessment of the Clean Lakes Program is in its incipient stage. In spite of its recency, the art of social impact analysis has progressed very rapidly over the past decade and a number of different methodologies have emerged to the benefit of Lake Restoration Evaluation. Over 50 different methodologies have hit the literature since 1964. Some of the more celebrated ones are (1) Battelle Environmental Evaluation System (Dee et al., 1972), (2) the Bureau of Reclamation's The Multiagency Task Force Method, (Bureau of Reclamation, Mississippi, 1972), (3) The Environmental Impact Center Method (Environmental Impact Center, 1973), (4) The Corps of Engineers' Valley Diversion Method (U.S. Army ED, 1976), and (5) The Soil Conservation Service's Guide to Environmental Assessment (Soil Conservation Service, 1974). Another is the Techcom Methodology developed and refined by Peterson et al., (1971) and now employed in a number of studies. An important recent synthesis is developed by Solomon et al., (WRAM) at Vicksburg, Mississippi (1977). Still, as one might predict, no one approach is generally accepted, even though a number of those available could substantially improve assessments.

Solomon et al., (1977) evaluation of eight different methodologies has revealed that none met all of his criteria for adequacy or completeness. He correctly points to the lack of measurement techniques and predictive technologies for many required variables in social impact analysis (Ibid., 1977: 18). The state of the art in social impact assessment, therefore, is diffused and demanding some theoretical integration. As Solomon et al. point out, appropriate methodologies must be (1) responsive to Principles and Standards, (2) comprehensive of all kinds of impacts, (3) dynamic enough to incorporate new variables and techniques, (4) sufficiently flexible to be applicable to various magnitudes and locales of development, (5) objective from either the standpoint of quantitative or subject data, (6) implementation in the field and with time or money constraints, and (7) replicable to the extent that others using the same framework would produce the same results in the same setting. Short of this we remain in a quandary.

CULTURAL ECOLOGY AS A METHOD FOR SOCIAL IMPACT ASSESSMENT

We regard the restoration of lakes as a cultural process. It involves more than just the application of a technology to a resource in order to modify it and thereby make it more immediately useful to groups of people. It also encompasses people's values, both pro and con, and their actions in order to finally employ an appropriate technology. As is the case in any technologically induced change of a resource, the new circumstance of the resource reciprocally feeds back upon human beings both in the immediate and peripheral setting. The more relevant the resource is in the first place, the more marked the impacts of its change. Social impacts normally are of a very broad nature. They filter through various kinds of institutions and ultimately affect people's attitudes and values in either direct or indirect ways.

The design and theoretical orientation for our social research into lake restoration at Liberty Lake are derived from a cultural-ecological model modified after Julian Steward (1955). Steward notes the utility of considering human adaptation and cultural development in terms of evolutionary processes. The evolutionary model makes explicit the relatedness of cultural and ecological systems whether they are part of a greater systematic linkage or are linked to each other in a causal or developmental manner. The field of cultural ecology derived from this orientation takes the linkage into account in terms of three fundamental procedures: (1) analysis of interrelationships of exploitive or productive technology and environment, (2) analysis of human behavioral patterns involved with the exploitation of a given area or resource, and (3) analysis of "the extent to which the behavior patterns entailed in exploiting the environment affect other aspects of culture" (Steward, 1955:40-41).

Implicit in Steward's design is the following type of relationship:

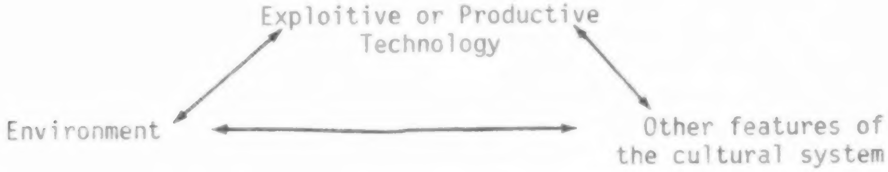

Figure 1. Cultural-Environmental Interrelationship

The fundamental linkage of the cultural system to environment, according to Steward, is the role of technology. Some technological features emerge as more important so far as cultural relatedness is concerned. Steward points out that the "relevant environmental features depend upon the culture: in that more developed cultures are less dependent upon the environment" (Ibid., 1955:40). Our own work (Hogg and Honey, 1975) has caused us to doubt this proposition of Steward. In fact, we have found that industrial-urban cultures are more intricately tied to features of environment. We will admit that Steward properly notes that a full grasp of the relationship between cultural and environmental systems can only be attained by a holistic examination of such factors as demography and settlement patterns, land use and tenure, and social structure, both in the past and present. To consider any of these separately runs the risk of failing to note their critical linkages. He correctly emphasizes that only by tracing the relevant history of a culture can we expect to understand its specific nature. An empirical rather than deductive method, therefore, is essential to the reconstruction out of which factors of form, function and sequence might be identified (Steward, 1955:18-19).

The determination of these features of a cultural system's interrelated behavior patterns, as these in turn relate to the environment, is the objective of cultural ecology. The manner in which technology is utilized by a cultural system and the extent to which an environment permits the use of a given technology will vary reciprocally. Cultural ecology, then, seeks to explain the origin of particular cultural features and patterns which characterize different areas.

Of further importance to this model is the concept of "cultural core," or central environmental feature. For the most part, a central environment feature can only be empirically determined and is usually associated with a long and involved cultural history. The immediate distinguishing significance of the central environmental feature is its interrelationships with primary cultural activities such as subsistence of economy. Examples include lakes, rivers, topographical features, flora and fauna. These clearly vary from one cultural context to another.

The appropriateness of Steward's work to social impact assessment of lake restoration projects, as at Liberty Lake, is seen through the notion of linkage of technological and environmental features to certain kinds of associated behavioral patterns, and then to other aspects of culture such as values and attitudes of people. These are linked in a specific way, one which fundamentally depends on the nature of technological-environmental relationships. Environment thus becomes an effective influence on culture, and provides an explanation of the origins of particular features and patterns of culture which characterize different areas. In this manner, then, cultural evolution can be attributed to new adaptations made by people as required by changing technologies and behaviors in relation to differing environments.

The application of Steward's theoretical framework to social impact assessment of a lake restoration prgram emerges in the form that is diagrammatically illustrated in Figure 2. The principal components of the design are as follows: 1) the historical emphasis serves to identify and explain the nature of the central environment feature (or the centrality of a particular feature) and its interrelationship with patterns of culture; and 2) the environmental-cultural system interplay determines to what extent the environment will permit or prohibit technological innovations; and, it identifies the special features of the cultural system on which adaptation of people depends.

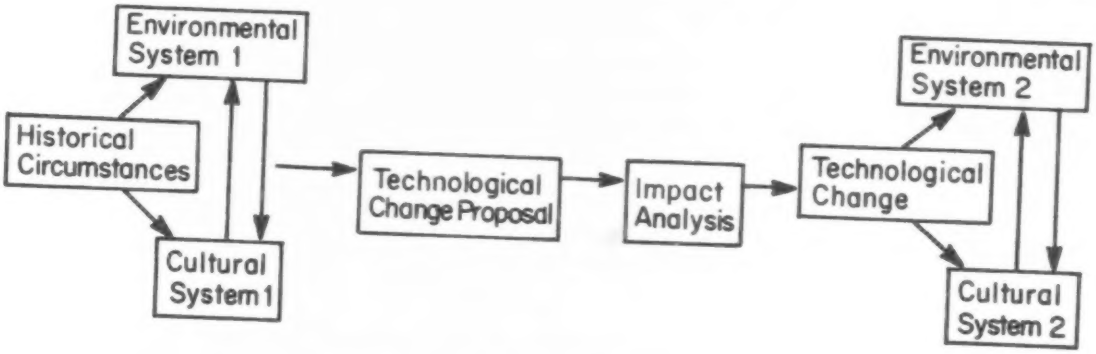

Figure 2. Cultural Ecology and Impact Analysis

The design possesses qualities of a "dynamic systems model" (cf. Fitzsimmons et al., 1975) in that it calls for the observation and analysis of related cultural components such as the economy, resource use and abuse, institutional involvements, socio-political process, and public attitudes. It thus allows for the conceptualization of the cultural-environmental circumstance. The application of the design is not restricted by the size or complexity of the project or its setting.

The application of the cultural ecology framework to social impact analysis establishes a comprehensive requirement for data and explanation not realized in many other methods of social research. This method demands historical, geographical, and ethnological information bases and requires a specification of their relationships. Properly employed, a cultural ecological study will show basic developmental patterns which have led to the present social circumstances of an area planned for subsequent development. Insofar as it specifies processes through time, it allows for intelligible projections of future circumstances based upon knowledge of definite cultural processes which operate in the present.

PRESENT STATE OF THE RESEARCH

Our data collection for our social impact research on Liberty Lake commenced with a comprehensive literature search of all pertinent materials extant on the Spokane Drainage Basin in eastern Washington. Information now is being assembled from public and private collections from various agencies, individuals, and institutions from the states of Washington and Oregon. Holdings of libraries, museums/historical societies, state and federal agencies and others, as appropriate are being included. Such information primarily is being selected from newspapers, journals, diaries and letters, books, periodicals, research documents, and any other written material that describes or explains the cultural-environmental circumstance of the Liberty Lake area. Emphasis here is on the history of the technological-environmental situation of Liberty Lake, the extent to which Liberty Lake has been a relevant resource (to whom and when.)

As an interim phase in our methodology, we now are in a field orientation/indoctrination period in conjunction with the literature collection phase. This consists of visits to the physical setting, establishing preliminary contacts and introductions with selected individuals and agencies involved in the rehabilitation effort. Field observation will allow for more precise formulation of specific hypotheses and for refinement of our analytic framework.

The two preceding steps are providing a basis for preparation of instruments for collection of quantitative information in the field. Quantitative data will emerge from two sources; they are interviews and written enumerative sources.

Structured interviews also will be conducted to collect quantitative data on social characteristics and attitudes of the population. They will be administered to a representative sample of the population adjacent to and in near proximity to Liberty Lake. Three samples will be selected. They are (1) recreational and other users, (2) residents of the immediate area impacted by restoration, and (3) residents of the secondary or adjacent area within Spokane Valley. Quantitative data collected on users will be drawn from individuals or groups participating in some recreational or commercial activity at the lake. These data will provide details of the present technology and the lake, as well as behaviors and attitudes pertaining to the lake. The primary impact area sample will be comprised of resident property owners, operators of commercial establishments, and/or agencies that are usually in close proximity to the lake and who will be more immediately, and directly, impacted by the restoration effort. Secondary respondents are those usually more geographically removed and less immediately affected (cf. Hogg and Honey, 1977). Data derived from this sample will allow measurement of the extent of Liberty Lake's centrality as a resource.

Unstructured interviews will serve to supplement data gathered for analysis of change in the use of Liberty Lake, the behaviors of people in reference to the lake, and the relationship of these to attitudes and values. The aforementioned literature review provides the primary base for the unstructured interviews. Unstructured interviews primarily will be aimed at key individuals who have a special or unique understanding or knowledge that relates to some facet of the research problem. Other unstructured interviews will be directed at various public and private agencies who also can assist in the information gathering process. Post coding of unstructured interview data will allow for quantification of some items.

Quantitative data will be subjected to computer analysis. They will be processed to develop population profiles through cross-tabulations and frequency distributions. Different types of statistical analysis will be accomplished, most likely including analysis of variance (ANOVA) and regression analysis to determine variable linkages. Non-quantitative data will be analyzed typologically for special interpretations of cultural process and function. Efforts in the research will primarily be directed toward maintaining a "qualitative-quantitative mix" (Pelto, 1970:44-45). Subjectively derived data will be employed to give additional interpretation of numerical data. All data will be analyzed in a manner which will provide an interpretation of the past, a profile for the present, and a reasonable projection of the future circumstances of Liberty Lake.

A "typical" evaluation study of lake restoration projects is impossible to specify since so few have yet been completed. An ideal type study should at least accomplish several major procedures (Honey and Hogg, 1978). We specify first that preliminary ethnographic and library research is essential for determining the cultural-ecological circumstance of the setting to receive the project. Second, a social profile is essential for any meaningful projections of "with and without project configurations" and for determining evolutionary trends. Third, the isolation of significant effects categories and effects, as evaluated by meaningful criteria of social and cultural functioning, is the final and most important step in the typical process. In our case, it allows for assessments of adaptation and maladaptation to environmental features. These, then, should be monitored to test the methodology employed.

Several levels of assessment may be sought in any project. First involves simply the identification of key social variables where impacts might reasonably be expected. Second includes an examination of interrelationships between variables in the social present in order to establish a baseline for subsequent future circumstance projections. Third, from these levels the research may accomplish projections of a "without project, with project, and with project alternatives" patterns from which comparisons can be made and significant effects and effect categories can be isolated. The research should specify at its onset the level of understanding sought. It should detail any socio-cultural omissions made. Our approach in this project is to attempt to achieve all three levels of analysis and to omit as little as possible.

Finally, we feel it should be noted that social impact assessment is far more than just basic social science research. Here the ethics of objectivity and social responsibility come head on in a serious circumstance where real people's life quality and social well-being must be carefully examined but treated in objective terms. We all live with biases, but these must be acknowledged and ignored so far as objective assessment is concerned.

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PROPOSED METHOD FOR EVALUATING THE EFFECTS OF RESTORING LAKES

by

D. B. Porcella*, S. A. Peterson**, and D. P. Larsen**

INTRODUCTION

An ongoing program to demonstrate methods for restoring polluted lakes and preventing pollution of clean lakes is being funded with EPA/local matching (50/50) money as directed by sections 314/104(h) (Clean Lakes) of PL92-500 (Federal Water Pollution Control Act Amendments of 1972). To aid in evaluating the efficacy of the various restoration techniques, comprehensive limnological evaluations are being conducted on a subset of lakes selected from all those being restored under the 314 program (Porcella and Peterson, 1977).

The evaluation grants are the outgrowth of several questions about lake management. Suppose the quality of a lake is perceived as needing protection or as being undesirable; can objective criteria be related to that perception? How does one change a lake to another specific condition or at least change its water quality? What are the effects of changes that occur in the watershed or in the lake on the water quality of the lake? How do various restoration techniques compare in terms of effectiveness?

Thus, the objectives of these detailed limnological evaluations of lake restoration projects are:

1) To determine the effectiveness of the specific restoration manipulation(s) at a given lake.

2) To compare the effectiveness of various restoration processes on different lakes.

The above questions and objectives reflect a need for predicting lake dynamics and future steady states as related to physical, chemical and biological factors and their interactions. Although it is probably not possible at this time to use sophisticated and precise means of predicting lake, biotic community, and specific organism responses to specific manipulations, it is

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necessary for managers to be able to predict manipulation effects on generalized variables that represent the more detailed and complex interactions of aquatic communities. Such "target" variables must be measured so that the effects of lake restoration projects can be evaluated and then the above questions answered. For the limnological evaluations discussed above, the target variables will be measured over a period of time extending from prior to the application of restoration (baseline) to a significant time after the restoration has been completed.

Two basic approaches will be used in achieving the evaluation objectives: 1) target variables will be based on the concept of nutrient balance similar to Vollenweider's analysis that began in the late 1960's (Vollenweider, 1968, 1976; Dillon and Rigler, 1974); 2) target variables will be selected to represent general lake water quality and combined in a logical fashion to provide an index number (Lake Evaluation Index, LEI).

Data appropriate for determining phosphorus and nitrogen loading of lakes and for estimating the LEI from the individual target variables will be used to compare lake quality before and after application of lake restoration methodology in each lake and to estimate the quantitative effects of the restoration on that specific lake. Then the effects of specific restoration methodology will be evaluated in terms of effects on external and internal loading and the predicted effect of that changed loading as compared to observed values in all lakes being evaluated. Similarly, calculated and observed effects on individual target variables and the LEI will be determined. The individual target variables that compose the LEI will be transformed to produce a scale of 0 to 100 so that comparisons can be made easily.

In this report we describe the basic concepts of lake quality evaluation and the data needed to perform the evaluation. In addition, we describe the concepts relating to the development of a LEI useful in performing the evaluation. We emphasize that we are presenting proposed methods; modifications and refinements no doubt will occur as our experience increases.

EVALUATION VARIABLES

BASIC APPROACHES

Many variables can be and have been measured in lakes; most measurements are fairly costly but results are not all of equal value in assessing lake quality. This is why it is necessary to develop concepts and approaches which limit the number of measurements. It is assumed that the Vollenweider Approach and the LEI are useful concepts for meeting the objectives of EPA's Clean Lakes evaluation program.

VOLLENWEIDER APPROACH

Considerable development of the phosphorus loading concept has occurred (see Vollenweider, 1976; Dillon and Rigler 1975; Lorenzen, et al., 1976; Lung et al., 1976; Larsen and Mercier, 1976; Chapra and Tarapchak, 1976); the necessary measurements are listed in Table 1. This approach seems reasonable because it is relatively simple, has feasible data requirements, considerable

TABLE 1. A LISTING OF MEASUREMENTS NECESSARY TO PERFORM ANALYSES OF LAKE ECOSYSTEMS USING NUTRIENT LOADING CONCEPTS (VOLLENWEIDER APPROACH).

Parameter	Water	Phosphorus	Nitrogen
depth area curves	X		
depth volume curves	X		
evaporation	X		
precipitation	X		
inflow (Q)	X		
outflow (Q)	X		
mean depth (maximum volume and area)	X		
inflow concentration*		X	X
in lake concentration*		X	X
sediment bulk concentrations and/or sediment release rates		X	X

* See Larsen, D. P., this publication pp. 311 for sampling protocol.

research has been done or is in progress, and external inputs are related to watershed activities and thus to possible control strategies.

LAKE EVALUATION INDEX

Various trophic state indices have been proposed (120 separate citations were reviewed by Shapiro, 1977; Uttormark and Wall, 1975; and Brezonik, 1976). The reviews conclude that there is no universal and completely satisfactory index of lake water quality. Generally, indices are designed for specific uses and for a set of regional or local lakes (Table 2). Ideally, a simple index of lake quality should be developed that 1) is not lake specific, i.e., it can be generalized to all lakes, 2) is related to all uses, and 3) is objective, independent of other variables, and easily measured. However, lakes are complex systems having many variables and their waters have many beneficial uses; at our present state of lake understanding an indicator(s) may be inadequate to satisfy all of the above criteria.

The difficulties in achieving these criteria can be seen in the variety of lake classification schemes shown in Table 2. Many critical reviews of the concepts and approaches for lake classification have been published but with little consensus (Bortleson, et al., 1974; Brezonik, 1976; Carlson, 1977a; Donaldson, 1969; Fruh et al., 1966; Hooper, 1969; Inhaber, 1976; Margalef, 1958; Shapiro, 1977; Sheldon, 1972; Stewart, 1976; USEPA, 1974; Uttormark, 1977; Vallentyne, et al., 1969). Some consensus can be gained from observing that the most commonly used variables include Secchi depth, DO, phosphorus, chlorophyll a, and nitrogen compounds.

In this discussion an LEI is proposed that incorporates a minimal set of limnological variables required to evaluate the limnological effects of lake

TABLE 2. THE CLASSIFICATION OF LAKES USING VARIABLES THAT REFLECT THEIR LIMNOLOGICAL CONDITIONS.

Reference	Dependent Variable	Number of Variables	TP	OP	TIN	TON	TN	POC	PP	CA	DO	pH	SD	Morph.	M/A	WQ	Ratios	Other Variables	
																		Floral	Faunal
Brinkhurst *et al.*, 1960	Trophic state	1																	X
Brook, 1965	Trophic state	1																X	
Carlson, 1977b	TSI	3	X							X			X						
Dillon & Rigler, 1975	Trophic state	3	X							X			X						
Dobson *et al.*, 1974	Trophic state	4	X					X		X			X						
Harkins, 1974	Standards SM	4	X								X	X				BOD$_5$ alk.			
Hayes, 1957; Hayes *et al.*, 1964	Fish PI	3												Mean depth, cm					
Hutchinson, 1938	Trophic state	1									X								
Jarnefelt, 1958	Trophic state	1																	X
Lueschow *et al.*, 1970	RTSI	5	(X)	(X)	X	X					X		X		Algae				
McColl, 1972	RTSI	9	X	X	X					X	X		X			Alk., Fe, Mn Light Trans.			
Megard *et al.*, 1978	Trophic state	1																	
Michalski & Conroy, 1972	Trophic state	6								X	X		X	Mean depth, Ryder			Fe/P		
Miller *et al.*, 1974	Trophic state	1													Bioassay				
Mortimer, 1941	Trophic state	1									X								
Neel, 1977	TSI	2												Residence Time			Mg/Ca		
Rawson, 1960	RTSI	8								X			X	Mean depth		Temp., TDS		Plankton	Benthos, fish
Reynoldson, 1958	Trophic state	1																	X
Rodhe, 1958, 1969	Trophic state	1							X										
Ryder, 1965	Fish production	2												X		TDS			
Sawyer, 1947	Trophic state	2		X	X														
Shannon & Brezonik, 1972	TSI	7	X			X			X	X			X			Cond.	Cation		
Skulberg, 1966	Trophic state	1													Bioassay				
Stockner, 1972	Trophic state	1																Sediment diatoms	
Toerien *et al.*, 1975	Trophic state	1													Bioassay				
USEPA, 1974	RTSI	6	X	X	X					X	X		X						
Uttormark & Wall, 1975	LCI	4									X		X		X	Fish kills			
Vollenweider, 1968	Trophic state	2	X				X												
Winner, 1972	RTSI	9							X	X			X		Seston	Assimilation	P/B, pigment	Diversity	Diversity
	Frequency of use		8	4	4	2	1	1	3	8	9	2	11	6	5	12	5	4	6

* Abbreviations: TP - total phosphorus, OP - orthophosphate P, TIN - total inorganic nitrogen, TON - total organic N, TN - total N, POC - particulate organic carbon, PP - primary productivity, CA - chlorophyll *a*, DO - dissolved oxygen, SD - Secchi depth, Morph. - various morphological parameters, M/A - macrophyte and other algal variables, WQ - other water quality variables.

restoration projects. (A discussion of data needs for phosphorus distribution in lake ecosystems is contained in the paper by Larsen, pp. 311 in this publication). The LEI is intended for a specific use although its generality may increase with application to other studies. The concept is simpler than the Vollenweider Approach, but data requirements are quite similar. The measurements relate to previous studies and in some cases conform to perception of lake problems and, therefore, to phenomena the general public can see.

Lake quality variables can be grouped roughly into hydrological, morphological, physical-chemical and biological types (Table 3). In most cases hydrology is not expected to be significantly affected by lake restoration. Changes in mixing patterns and residence times will occur in dilution/flushing projects; mixing can result from some dredging, aeration, and other projects. Some morphological variables will be greatly affected; depth and volume will be changed by dredging and/or outlet structure changes and diversions. Most changes will be seen in terms of physical-chemical interactions (nutrients, other salts, light and temperature) and biological responses to these changes (flora, fauna and dissolved oxygen). Measurement of all factors related to these changes is impractical. Thus it is necessary to select target variables that indicate general water quality.

TABLE 3. CATEGORIES OF MAJOR LAKE RESPONSE PARAMETERS

Hydrologic	Morphologic
Inflows	Shoreline shape
Evaporation-transpiration	Mean depth
Precipitation	Area
Outflows	Volume
Mixing	
Residence time	
Physico-chemical	**Biological Response**
Phosphorus	Chlorophyll a
Nitrogen	Secchi disk
Iron	Macrophyte biomass
Trace Metals	Faunal densities
Carbon	Phytoplankton parameters
Cation/Anions	Dissolved oxygen
pH	
Light*	
Temperature*	

* affect nutrients and growth responses

SELECTION OF TARGET VARIABLES FOR LEI

The basis for the LEI concept is that lake water quality problems are defined as being caused largely by or associated with increased nutrient concentrations in the lake. In most cases, phosphorus will be the nutrient of concern (Bartsch, 1972; Porcella et al., 1974; Schindler, 1977). For example, in Figure 1, a sequence of cause and effect events are shown which would occur under conditions where phosphorus was limiting. An increase in lake phosphorus concentration would cause an increase in primary productivity as measured by chlorophyll a. Simultaneously there would be decreases in Secchi depth (higher turbidity) and hypolimnetic DO (higher BOD from algal growth, i.e., respiration exceeds production). For this scenario, data on Total P (TP), chlorophyll a (CA), Secchi depth (SD), and DO can be used to express changes in lake water quality.

This scheme applies when phosphorus limits primary production of phytoplankton but not when nitrogen is limiting (USEPA, 1974; Miller et al., 1974). Thus it is necessary to measure nitrogen compounds (total nitrogen, TN) as well. Other nutrients can limit primary production of lakes (Goldman, 1965); such an occurrence is infrequent relative to phosphorus and nitrogen limitation and thus those factors will not be included in order to maintain the concept of measuring only the essential variables for developing a reliable LEI.

Macrophytes (MAC) are an important part of lake primary production that are not measured by phytoplankton chlorophyll a or most primary productivity methods and yet have significant effects on nutrients, DO, SD and other water quality parameters and lake beneficial uses. Whenever lakes are relatively deep, primary productivity is dominated by phytoplankton. Sedimentation in lakes decreases lake depth (whether sedimentation is due to settling of organic materials produced in or out of the lake or is due to settling of inorganic materials). Also some lakes are naturally shallow for topographical or geological reasons. In all cases lakes having shallow zones (<6 meter depth contours) usually develop significant macrophyte growth when nutrients are available. Therefore data on macrophytes are required also.

The evaluation phase has been designed to analyze productivity problems. However, lake water quality problems resulting from BOD inputs and suspended solids loadings would affect the target variables DO and SD, also. Additional analyses would be required to determine effects of lake restoration on bacteriological problems as would be required for other non-eutrophication related problems (toxicity, oil spill, salinity). Although investigations at specific lakes need be concerned with those problems where relevant, time and dollar constraints confine this overall evaluation phase to those limnological variables related chiefly to eutrophication problems.

DEFINITION OF TARGET VARIABLES OF THE LEI

The above rationale suggests that the following target variables are sufficient for the purposes of developing an LEI:

1) Secchi depth (SD)
2) Total phosphorus (TP)
3) Total nitrogen (TN)
4) Chlorophyll a (CA)
5) Dissolved oxygen (DO)
6) Macrophytes (MAC)

These target variables have the following attributes: 1) They span most of the major water quality problems that affect uses of lakes, 2) they duplicate

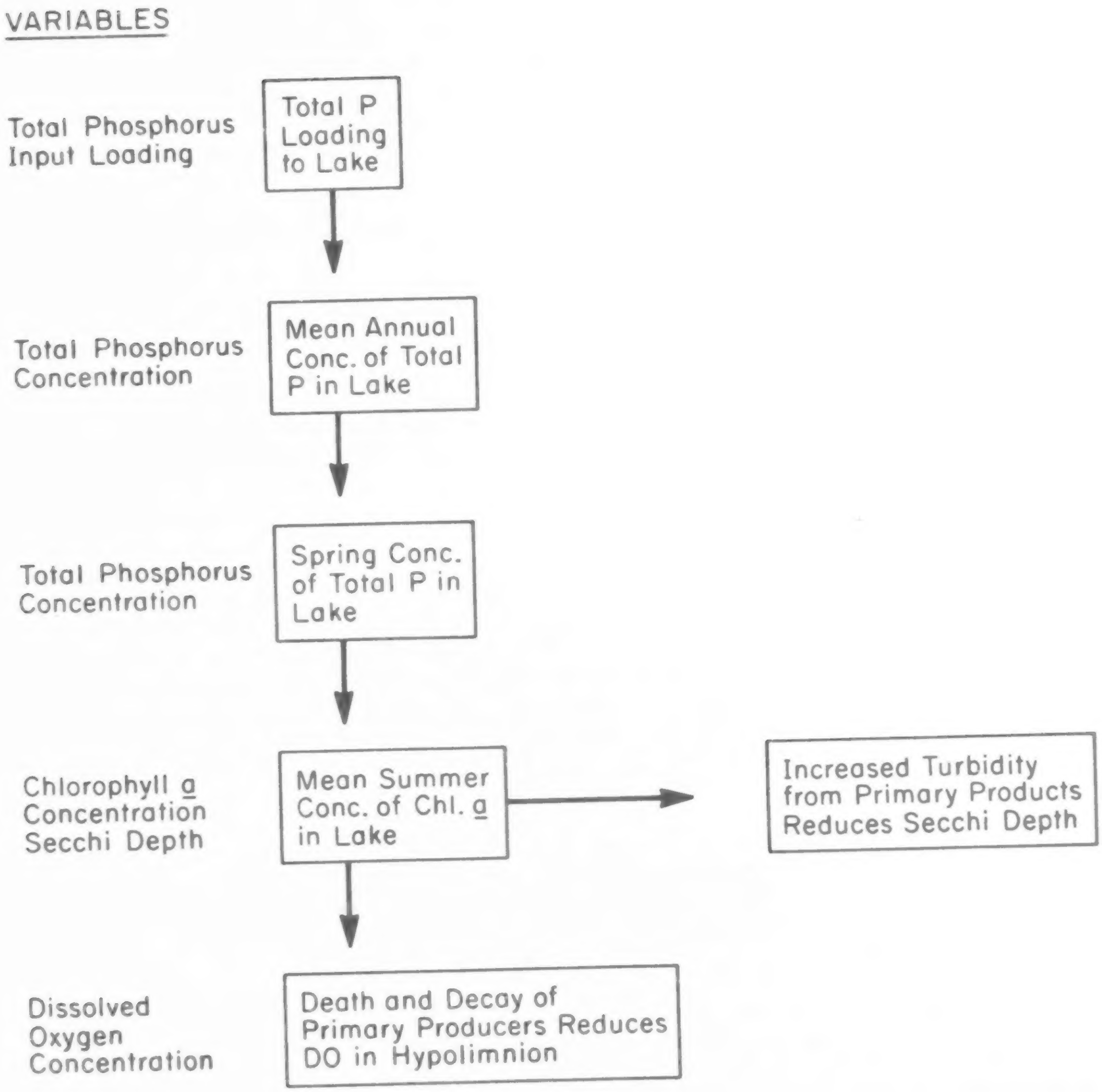

VARIABLES
Total Phosphorus Input Loading
Total P Loading to Lake
Total Phosphorus Concentration
Mean Annual Conc. of Total P in Lake
Total Phosphorus Concentration
Spring Conc. of Total P in Lake
Chlorophyll a Concentration Secchi Depth
Mean Summer Conc. of Chl. a in Lake
Increased Turbidity from Primary Products Reduces Secchi Depth
Dissolved Oxygen Concentration
Death and Decay of Primary Producers Reduces DO in Hypolimnion

Figure 1. Conceptual sequence of cause and effect relationships in lake eutrophication processes (modified from Chapra and Tarapchak, 1976).

most of the parameters contained in other indices and in the Vollenweider Approach and 3) they are commonly and for the most part, relatively simply measured.

However, the target variables are not mutually exclusive, independent variables. They are interrelated in complex ways. They may be additive, as may be the case for MAC and CA. In other cases there may be a concentration dependent maximum in reference to one variable (CA and DO) and a relatively linear relation in reference to another (CA and TP). Because the LEI is a composite of variables that in specific cases or at different seasons can be unrelated, negatively related, or positively related, interpretability of the LEI will probably be limited. However, the range of lake types to be evaluated will be broader.

Having selected the above target variables, other questions arise:

-When, where and with what frequency are they measured?
-Are other data needed to calculate the target variables?
-Are other data needed to support the development of an LEI?

The following sections provide some answers to these questions. Sampling needs and concepts, previous work on each target variable, and other data requirements are discussed. Methods of analysis are specified in Appendix A.

SAMPLING

For practical reasons, funding will limit the frequency and density of sampling. During critical flow periods (spring runoff, summer low flows) and the growing season (periods of high primary productivity), sampling should be at least biweekly and weekly if possible. Overlap of these periods provides some sampling economy. For uniformity the July-August period is specified for the target variables as used in the LEI. At other periods monthly sampling should be adequate. Generally, time of day is very important and lake measurements should be restricted to 1000 hrs to 1400 hrs standard time, preferably closer to 1200 hrs.

At least 90% of the tributary inflow should be determined by measurements, continuously if feasible (USEPA, 1975a). Estimates of runoff (USEPA, 1975a) and groundwater input (Lee, 1977) should be obtained and their significance to loading assessed to determine if more accurate measurement is necessary (USEPA, 1974).

Sampling for chemical analysis should allow estimation of the total lake loading of TP and TN and in-lake mass at a point in time for TP, TN, DO, and CA. Sampling of the major lake basins and littoral zones should be as judged appropriate by the investigator. Vertical profiles of the variables should be determined at least by a bottom, midpoint and near surface sample at the deep station(s).

Variables such as chlorophyll a, open water primary productivity, and nutrients relate principally to measurements made in the water column in the epilimnetic zone (defined as being the layer enclosed between the water surface and the lake bottom or water depth at the thermocline).

For the LEI, mean epilimnetic zone nutrient and CA concentrations will be used. Mean upper level concentrations (such as 10 meter depth, etc.), photic zone, maximum epilimnetic concentrations, and maximum lake concentrations could be used if necessary. Although loadings, total lake mass (kg/lake), and areal measurements ($mg \cdot m^{-2}$) are useful concepts, they are not used herein for the LEI because such measurements vary greatly and independently with drainage basin, lake volume, area, and residence time. They may be combined at some future time since data can be normalized using various loading equations to relate lakes of differing morphology (Allum, et al., 1977).

SECCHI DEPTH

The depth of light penetration into lakes is controlled by the sun and climate, season, water color and turbidity (Tyler, 1968). Light controls photosynthesis hence primary production in the lake, defining zones that limit the depth of phytoplankton net production and the distribution of macrophytes. The SD is a common and simple method for estimating the maximum depth of light penetration in lake waters. SD needs to be measured at the same time (near noon) as other variables and as often as the lake is sampled; however, SD is measured only at the deepest sampling station.

Because we expect SD to estimate the limit to light penetration during the growing season, the target variable will be the mean SD during the months of July and August. During this period SD will vary chiefly according to the concentration of phytoplankton. Because the highly colored waters of certain lakes affect SD (Shannon and Brezonik, 1972), it may become necessary to separate lakes into classes or types. Although classification will be avoided if possible, color should be noted where observed.

TOTAL PHOSPHORUS

Sawyer (1947, 1966) was the first to rate eutrophication levels based on nutrient concentrations; inorganic phosphorus concentrations of 10 $mg \cdot m^{-3}$ when vertically uniform concentrations exist was defined as the threshold above which nuisance algal blooms could be expected to occur. Vollenweider (1968) used Sawyer's estimate to define eutrophic conditions by relating total phosphorus in the spring and summer to annual loading rates from inflows. More sophisticated mass balance models consider sediment loading (Lorenzen et al., 1976; Lung et al., 1976; Larsen and Mercier, 1976; Vollenweider, 1976) and attempts are now being made to define the role of the different forms of phosphorus. For example, TP includes non-algal P and would introduce some error in interpretation. For purposes of the LEI, summer (July and August) total P (TP) concentration averaged through the epilimnetic zone will be used.

TOTAL NITROGEN

A value of 300 $mg \cdot m^{-3}$ of total inorganic nitrogen similar in concept to phosphorus, was defined by Sawyer (1947, 1966) to relate to incipient eutrophication problems. Leuschow et al. (1970) proposed that total inorganic and total organic N be used as variables for lake characterization. Because total N would include inorganic nitrogen forms (potential growth of primary producers) as well as particulate nitrogen (primary producers), it was chosen as the lake target variable. Unfortunately total N also includes detrital N and soluble organic forms of N that might or might not be available for growth; in addition there is considerable analytical error in the Kjeldahl measurement and this plus its difficulty and cost often lead to its exclusion as a measured parameter. For purposes of the LEI, the summer (July and August) TN concentration averaged through the epilimnetic zone will be used even though these disadvantages exist.

CHLOROPHYLL A

Many investigators have correlated chlorophyll a and phosphorus loading and thereby related a level of eutrophication to chlorophyll levels (NAS-NAE, 1973; Jones and Bachmann, 1976; Dillon and Rigler, 1974; Porcella *et al.*, 1974; USEPA, 1974). These approaches have a logarithmic functional relationship in common; thus, loss of beneficial use occurs with increasing chlorophyll a concentrations, but detriment increases more rapidly at low concentrations and less rapidly at higher concentrations.

Dobson (1974) defined chlorophyll a as a function of clarity using the inverse of SD in meters: $CA = 1.14 (30/\overline{SD})$. Similarly, a non-linear approach was used by Carlson (1977b): $\ln SD = 2.04 - 0.68 \ln (CA)$.

Because CA concentrations are a function of other variables and like SD, can be related to a perception of the quality of a lake system, epilimnetic zone concentrations can be used to define levels of quality for the other target variables. For this reason it is an important variable. However, CA does not provide the dimension of the composition of the phytoplankton population. Consequently, it is necessary to characterize the algal species comprising the phytoplankton community. To minimize effort and costs associated with this task, only the three dominant genera and their numerical concentration in a single epilimnetic zone composite sample collected in conjunction with the CA sample need to be determined. The CA target variable is defined as were TP and TN: the summer (July and August) CA concentration averaged through the epilimnetic zone.

DISSOLVED OXYGEN

Several approaches have been used for analyzing DO data: hypolimnetic DO has been used to characterize lake trophic status (Uttormark and Wall, 1975); DO deficit (Hutchinson, 1938) and deficit rates (Mortimer, 1941) have been suggested (Hutchinson, 1957); hypolimnetic concentrations (Lueschow, *et al.*, 1970; Michalski and Conroy, 1972), a transformed minimum DO (USEPA, 1974), and DO concentration (Harkins, 1974) have been used.

These approaches all have disadvantages. Hypolimnetic DO represents a water layer which has a continuous demand due to heterotrophic breakdown of organic matter (excess production) but little or no replenishment from other sources (atmosphere, diffusion processes, inflow, primary productivity). Almost all of the demand for hypolimnetic DO comes from organic material settling through the hypolimnion or contained in the sediments (Lasenby, 1975). Consequently, oxygen demand by the sediments represents previous history of the lake system and changes in lake nutrient and productivity status may not be reflected in a change in DO demand without a significant time lag. Significant changes in nutrient inflow that are applied over a long period of time and/or changes in existing sediment chemical composition would be required before hypolimnetic DO patterns would be significantly affected.

Epilimnetic DO increases during the day due to photosynthesis and decreases at night from respiration. Sampling times must be uniform or, preferably, determined over diel cycles.

Total lake DO is the sum of these two layers and 1) could exceed calculated temperature limited equilibrium DO levels if photosynthesis is relatively high or 2) fall short of the saturation levels where respiration is relatively high. Ideally hypolimnetic DO would be the most useful indicator of respiration and respiration would be relatively independent of time and space effects on sampling. Unfortunately the volume of the hypolimnion of many lakes, particularly the lakes being restored in the Clean Lakes program, is small relative to the lake bottom area or is nonexistent. Thus many of the approaches described in the literature cannot be used due to morphological differences in lakes (Lasenby, 1975).

As a first step in developing a target variable based on DO, we have assumed that it is possible to estimate the instantaneous total lake equilibrium DO (EDO, mg/lake) from atmospheric pressure and the temperature-depth profiles. This value (EDO) is defined as the reference value for a clean water lake. A comparison of this value with the calculated instantaneous total lake DO (CDO, kg/lake) allows analysis of the relative quality of the lake ecosystem with respect to physical processes and respiration/photosynthesis. However, in highly productive lakes that stratify during the summer, DO supersaturation can occur in the surface waters while zero DO or undersaturation occurs in the bottom waters. Addition of these quantities (a positive and a negative) to obtain total DO could result in essentially no difference in comparison with EDO. Thus, for analysis of DO the incremented absolute values of the net difference with depth between EDO and CDO will be utilized to evaluate lakes:

$$\text{net DO} = \frac{1}{V} \sum_{i=0}^{i=ZM} \left| (\text{EDO} - \text{CDO})_i \right| \Delta V_i$$

where ZM is maximum depth, ΔV is the volume at a selected and convenient depth increment, and i is the increment. Determination of EDO and CDO would require measurement of DO and temperature profiles with depth at sufficient sampling sites to estimate total lake DO. Measurements should be based on average summer (July and August) values. Significant ($\leq 5\%$) inflow/outflow or volume changes would require adjustment of EDO estimates.

MACROPHYTES

So far we have defined variables that relate principally to the pelagic area of lakes, i.e., the deeper zone of open waters. Most eutrophic lakes are relatively shallow, but even deep lakes have shallow regions (the littoral zone) typical of neither the pelagic zone nor the drainage basin (watershed) which nourishes the lake. The littoral zone contains macrophytes which mark the transition from rooted upland or terrestrial producer organisms to planktonic producers of the open water ecosystem. Because macrophytes have not been used to a great extent for lake indexing, we present more background information on macrophytes than the other target variables.

Our concern is to develop a relationship between macrophyte biomass and nutrient variables (water concentrations, sediment concentrations, loadings) within the littoral zone because macrophyte problems occur in approximately 1/3 of the funded Clean Lakes demonstration projects. Generally, we define that high quality lakes have few macrophytes and lower quality lakes have more macrophytes in the littoral zone. Kettelle and Uttormark (1971) listed more than 40% of U.S. problem lakes as having macrophyte problems. Uttormark and Wall (1975) indicated that more than 20% of all the lakes they surveyed in Wisconsin had observable macrophytes and 40% of their problem lakes (Lake Condition Index $\geq$10) had severe macrophyte problems. Also, macrophyte productivity in the littoral zone can be a major fraction of organic matter to the lake system (Wetzel, 1975) and may be a significant source of nutrients as well (Howard-Williams and Lenton, 1975; Klopatek, 1975; Cooke and Kennedy, 1977). Hence, characterization of macrophytes generally is necessary to assess eutrophication processes in most lakes in addition to the analysis of watershed and open water processes and, for the lake restoration program, to estimate littoral zone areal distribution and biomass of macrophytes.

Macrophytes (algae, mosses, and vascular plants or weeds) may be attached emergent, submerged, submerged with floating leaves, or free-floating forms. These plants obtain nutrients in part from the water but also from the bottom sediments where many are anchored; thus they mark a second interface within the lake ecosystem, that between the lake bottom and the water. Among other physiological differences, vascular plants differ from algae and mosses because they are sensitive to pressure, probably because of the presence of gas containing tissues necessary for maintenance of the life cycle (Wetzel, 1975), and are thus physiologically depth limited to no more than approximately 10 meters. Most are limited to much shallower depths due to light attenuation.

The littoral zone can be divided into three different regions on the basis of macrophyte distribution zones (zones slightly modified from Hutchinson, 1975; Wetzel, 1975): 1) shallow zone ($\leq$1 meter deep); emergent, rooted macrophytes; these include swamps, marshes, and shallows, and can be classified as wetlands; 2) mid zone (1 to 3 meters); floating leaf vegetation ("usually are perennials that are firmly rooted with extensive rhizome systems"; Wetzel, 1975; p. 335); 3) deep zone (0.5 to 10 meters for weeds and deeper to the limits of the photic zone for mosses and macroalgae). Macrophyte dynamics include community and nutrient interactions and flux with time and among water, sediment and biotic components. Wetlands (shallow zone macrophytes) are excluded from evaluation because we are interested principally in the response of definable lake systems to restoration (Hutchinson, 1975). However, significant inputs of materials from wetlands should be assessed, if possible. The lake area itself will still extend to the typical boundary of the lake margin (water-land interface). Wetlands in freshwater ecosystems are defined as the area enclosed by the emergent (throughout most of life cycle), rooted, aquatic vegetation line on the deepening slope and by the line on the upland slope where vegetation requires saturated soils for growth and reproduction (Federal Register: 40/173: 41297, September 5, 1975).

Evaluation of macrophytes excludes wetlands but considers that the biomass of a lake ecosystem is the result of allochthonous inputs (tributaries, wetlands, direct runoff from terrestrial systems) and autochthonous plant growth (macrophytes, benthic and attached algae, and phytoplankton). Defining the area of macrophyte growth is the first step in developing an approach for evaluating macrophyte productivity.

The distribution of macrophytes in a lake, where no other growth requirements are limiting, is controlled largely by light. Consequently, denser macrophyte growth occurs on the surfaces of water by essentially free-floating plants (water hyacinth, duckweed) and throughout littoral zones of lakes where rooted plants receive sufficient light. Turbidity derived from allochthonous particles, turbulence, or due to phytoplankton or self-shading, can decrease light penetration and reduce the depth to which macrophyte communities extend. Thus, watershed activities or eutrophication effects which cause increases in phytoplankton may cause changes in macrophyte density. It may be feasible to relate SD to vascular plant distribution limits, because of its relationship to light penetration (e.g., 2 times SD; Dillon and Rigler, 1974; 2-5 times SD; Mackenthun, 1969; p. 30). Thus, the vascular plants of interest for evaluation are restricted to the littoral zone bounded by the shoreline, wetland or an upper limit maintained by mechanical disruption of life cycles by wave action or shearing by ice and bounded in deeper water by pressure or light limitation (e.g., defined by mean SD during the growing season, July-August).

The variables for relating macrophyte populations to nutrients and lake condition are obviously complex and interrelated with other variables. For example, it is possible for macrophyte problems to occur in lakes that have no algal blooms and vice versa. This complication arises because of differing nutrient sources and interaction with the phytoplankton community. Macrophytes obtain nutrients directly from the water column and from the lake sediments (via the roots) but phytoplankton obtain nutrients only from the water column. Also, in contrast to algal communities, primary production in macrophyte communities might be limited by nitrogen. There is no documentation of aquatic macrophytes' ability to fix nitrogen as occurs with terrestrial legumes or in lakes with heterocystous blue-green algae. Also, development of shallow water zones will be increased by the presence of macrophytes due to increased siltation rates as a result of their dampening effect on water velocities in specific areas of lakes.

Population density of macrophytes may or may not be related to changes in Secchi depth or DO and needs to be estimated as a separate parameter. Because of the dearth of management-oriented information on macrophytes, an arbitrary approach has been taken for macrophytes; based on experiences of the State of

Minnesota (Jessen and Lound, 1962) and the State of Wisconsin (Dunst, Wisconsin Dept. Natural Resources, 1976; personal communication), the following parameters of macrophyte communities have been defined as requiring measurement: Species present, density (plants/unit area), percent of lake surface area covered, water depth and substrate type for the type of plant, percent of theoretically available substrate determined on the basis of the 10 meter contour line or the light-limited macrophyte growth contours, whichever is least. The specific approach has been prepared in step-by-step fashion in Appendix B.

Using this approach and obtaining synoptic data from a large number of lakes, several hypotheses that relate macrophyte distribution and population density to light (turbidity) and nutrients in sediments and/or the water column could be tested:

1) total macrophyte biomass and/or density is related to light input and nutrient availability;

2) the light-limited distribution of macrophytes is a function of a Secchi Depth parameter;

3) the composition of nutrients is very important to macrophyte successional sequences;

4) nutrient availability is governed by sediment interstitial water and/or water column nutrient concentrations;

5) total macrophyte biomass in a lake system is relatively independent of the species composition and diversity;

6) the transport of sediment interstitial water nutrients into the water column by macrophytes is dependent on water column concentrations primarily and sediment concentrations secondarily.

In summary, the analysis of aquatic vascular plants (predominantly angiosperms) would allow 1) evaluating lake restoration techniques and 2) assessing whether certain functional relationships exist between light/nutrients and macrophytes. Thus, the study of macrophyte distribution is expected to integrate the effects and interactions of many environmental variables on macrophyte growth and distribution, namely light (season, latitude and turbidity), nutrients (sediment, water sources and nutrient type), and other variables (bottom substrate type, cations and anions, temperature, toxicants).

OTHER REQUIRED DATA

In addition to the above target variables and data required for the Vollenweider Approach (Table 1), the following data are required for all lakes although they are not part of any specific lake index (rather they are independent variables): area and volume relationships with depth, pH and temperature depth profiles with time, identification of 3 predominant algal genera and macrophyte genera, and total macrophyte biomass. This is a minimum program for evaluation.

Other analyses of specific research interest and specific to the particular problems of the region or locality are not precluded and in fact attention to those problems is needed. The data needs described in this report are designed to meet the overall objectives of the limnological evaluation projects and have been selected to minimize duplication and unnecessary additional work; thus, the collection of these data should not preclude other methods of evaluating lakes. In addition it may be important to establish relationships between certain variables as necessary for the specific problems of a specific lake. For example, measurements of SD relate closely to other variables, particularly turbidity. Suspended solids (SS, $mg \cdot l^{-1}$) relate to turbidity and hence SD. If inorganic solids settle out and are not resuspended, SS and SD relate to phytoplankton biomass and should have some relationship to chlorophyll a and primary productivity. Where data are available, primary productivity ($mg \cdot cm^{-2} \cdot yr^{-1}$) and algal biomass estimates ($mg \cdot l^{-1}$) from cell counts are useful for developing relationships to chlorophyll a concentrations.

TRANSFORMATION OF VARIABLES

In order to use the target variables in a meaningful way, they must be transformed to represent a scalar quantity which represents a value judgment (e.g., good or bad, best or worst, beneficial or damaging, etc.). The range of true scalar quantity must be the same for all target variables to allow variables to be combined and to allow comparisons between variables for different restoration projects. The approach suggested here derives from Carlson's (1977b) method of transforming SD data.

Carlson's approach was to take the greatest and least expected values for SD and assign a rating scale of 0 to 100. Then the functional shape of the curve relating the ratings was described mathematically. Conceptually, this can be accomplished for SD where 100 can be assigned to essentially no light transmission, 0 can be assigned to the light transmission of pure water and the Beer-Lambert Law can be assumed to apply to the functional relationship for the ratings. For the other variables the functional relationships that transform measured values to rating values are not as clearly defined even though a reasonable range of minimal (0) to maximal (100) impact can be assigned. These other variables are not as simply related to a single target as SD is for clarity (light penetration). Carlson avoided this problem in part by relating TP and CA to SD.

SECCHI DEPTH (SD)

To relate lake trophic state to a measurable variable, Carlson assumed that light intensity (I/I_o) as measured by Secchi disk disappearance decreases with depth (Z) according to the Beer-Lambert Law:

$$\ln I/I_o = -nZ$$

Increased turbidity owing to phytoplankton and other suspended material would increase the value of the extinction coefficient (n) and cause the disk to disappear at shallower depths. Carlson felt that using a logarithmic base of

2 instead of the natural logarithm would be more useful for translating the rating to the public.

Using a maximum limit for SD of 64 m (41.6 m was the maximum reported in Hutchinson, 1957; 43.25 m was reported for Lake Tahoe by Goldman, 1974), Carlson developed a trophic state index (TSI) for SD:

$$TSI = 10(\log_2 (64) - \log_2(SD))$$
$$TSI = 10\ (6 - \log_2 (SD))$$

Carlson's TSI was equated with the scalar rating value, XSD, for use in the LEI as follows:

$$XSD = 60 - 14.427 \ln (SD)$$
$$\text{and } XSD \leq 100.$$

(All rating values are confined to the range 0 to 100 to prevent undue weighting of single variables on the LEI.) Comparison of SD and the rating values are shown in Table 4.

Carlson's relationships for SD are based on surface concentrations of TP and of CA. The LEI target variables are defined for epilimnetic zone concentrations. Different slope values would be expected for rating curves of epilimnetic zone concentrations as compared to surface samples. Correlation of TP and CA using surface samples (Carlson, 1977b) or epilimnetic zone samples (Dillon and Rigler, 1974) indicated that differences in equation coefficients were minimal. Although SD coefficients could vary significantly, Carlson's will be used for the target variables in the LEI as defined for the photic zone concentrations, until new data show that significant differences occur.

TOTAL PHOSPHORUS (TP)

The shape of the curve relating the scalar value (XTP) to TP July-August average epilimnetic zone concentration is based on relationships between chlorophyll *a* and TP (e.g., Carlson, 1977b; Dillon and Rigler, 1974; Jones and Bachmann, 1976) and chlorophyll *a* and SD (Edmondson, 1972; Carlson, 1977b). These relationships suggest that TP is logarithmically related to the quality of lake water; higher TP results in greater algal populations and lesser transparency but the impact of the rate of concentration increase is less at higher concentrations.

The limits of Carlson's scalar values include the lower TP measurements but the higher values ($\geq$ 768 $mg \cdot m^{-3}$) must be defined as equal to 100. Oligotrophic lakes have lower values on the order of 1 $mg \cdot m^{-3}$ (Waldo Lake, OR; Malueg *et al*., 1972; Lake Tahoe, CA-NV, 0.9 $mg \cdot m^{-3}$; Goldman, 1974). Eutrophic lakes exhibit a wide range of maximum values of springtime orthophosphate P or summer TP (150 $mg \cdot mg^{-3}$, Jones and Bachmann, 1976; 330 $mg \cdot m^{-3}$, Miller *et al*., 1974; up to 3660 $mg \cdot m^{-3}$ of median TP, USEPA, 1974).

TABLE 4. RATING SCALE FOR LAKE WATER QUALITY PARAMETERS.

Rating (X)	Secchi Depth meters	Total P $mg \cdot m^{-3}$	Total N $mg \cdot m^{-3}$	Chlorophyll a $mg \cdot m^{-3}$	Net DO $mg \cdot l^{-1}$	Macrophytes % available lake area covered
0 (minimally impacted)	64.	0.75	5.2	0.04	0.0	0
10	32.	1.5	10	0.12	1.0	10
20	16.	3.0	21	0.4	2.0	20
30	8.0	6.0	42	0.94	3.0	30
40	4.0	12	83	2.6	4.0	40
50	2.0	24	170	6.4	5.0	50
60	1.0	48	330	20.0	6.0	60
70	0.50	96	670	56.0	7.0	70
80	0.25	190	1300	150.0	8.0	80
90	0.125	380	2700	430.0	9.0	90
100 (maximally impacted)	$\leq$0.062	$\geq$770	$\geq$5300.	1200.0	$\geq$10.0	100

Carlson's equation transformed for use in the LEI is:

$$XTP = 4.15 + 14.427 \ln TP$$
$$\text{and } XTP \leq 100$$

TOTAL NITROGEN (TN)

The formulation of TN July-August average epilimnetic zone concentration in the rating scale (XTN) was determined in relation to TP. The N/P ratios for phytoplankton average about 16/1 (mole/mole) or 7/1 (weight/weight) (Bartsch, 1972; Stumm and Morgan, 1970; p. 429). Thus TN is equivalent to 7.0 TP and the scalar rating is:

$$XTN = 14.427 \ln TN - 23.8$$
$$\text{and } XTN \leq 100.$$

Equivalent values of XTN and TN are shown in Table 4.

Oligotrophic lakes show values of TIN (TN data incomplete) to be about 1-50 $mg \cdot m^{-3}$ (Waldo Lake, OR, 50 mg $N \cdot m^{-3}$, Malueg _et al_, 1972; Lake Tahoe, CA-NV, 0.9 mg $N \cdot m^{-3}$ (NH_4^+ assumed to be 0.0); Goldman, 1974). Values for eutrophic lakes vary widely (710 mg $TIN \cdot m^{-3}$, Miller _et al_., 1974; median value 7355 mg $TIN \cdot m^{-3}$, USEPA, 1974), and XTN must be restricted to 100 when including values greater than 5330 $mg \cdot m^{-3}$.

CHLOROPHYLL _A_ (CA)

The average July-August epilimnetic zone concentration of chlorophyll _a_ (corrected for pheophytin; see Holm-Hansen _et al_., 1965; APHA, 1975 for analytical methods and Fee, 1976 for discussion of sampling problems for chlorophyll _a_) is related to the scalar value (XCA) as a logarithmic function as was discussed for TP, above. Concentrations of chlorophyll _a_ in oligotrophic and eutrophic lakes are encompassed by Carlson's equation (Table 5):

$$XCA = 30.6 - 9.81 \ln (CA)$$
$$\text{and } XCA \leq 100.$$

Equivalent values of XCA and CA are listed in Table 4.

DISSOLVED OXYGEN (DO)

The net DO calculated as an average over the principal summer months (July-August) is based on what the DO would be in a pure water lake and what is actually measured (see section on defining target variables). Without contrary information the scalar value (XDO) is assumed to be a linear function of the net DO. The best situation (XDO = 0) would occur if net DO was zero, and a very poor quality ($XDO \geq 100$) would exist if net DO is ≥ 10:

$$XDO = 10 \text{ (net DO)}$$
$$\text{and } XDO \leq 100.$$

Equivalent values of XDO and average net DO are listed in Table 4.

TABLE 5. SOME MAXIMUM AND MINIMUM CHLOROPHYLL A VALUES MEASURED IN LAKES.

Reference	System or Lake	Chlorophyll *a**, mg·m^{-3} Maximum Values	Chlorophyll *a**, mg·m^{-3} Minimum Values
Dobson, *et al*., 1974	Great Lakes	25.4	0.4
Jones & Bachmann, 1976	16 Iowa Lakes	262.2	6.8
" " " "	" " " and compiled data**	400.0	0.3
USEPA, 1974	209 lakes in National Eutrophication Survey	381.0	1.0
Winner, 1972	5 Colorado Lakes	34.1	1.0
Shannon & Brezonik, 1972	55 Florida Lakes (mean)	39.1	1.8
Fee, 1976	ELA lakes	327	<1.0**
Malueg, *et al*., 1972	Waldo Lake, OR	1.64	0.13 (Mean, 0-60 m deep)
Holm-Hansen, 1976	Lake Tahoe, CA-NV	<1.0	0.1
Extremes		400	0.1

* not all data corrected for pheophytin *a*

** estimated from graphical data

MACROPHYTES (MAC)

The area of the lake subject to growth of macrophytes can be defined as the area encompassed by the lake margin and either the 10 m line or the depth at which light becomes limiting to vascular plant distribution and growth (2 times SD) whichever is shallower. The percent of this area that is actually covered by vascular plants is defined as the target variable. Only relatively crude surveys during the growing season (July-August) are needed to assess the percent of that area that is actually covered by the vascular plants. The target variable, percent macrophyte area covered (PMAC), could be assessed in terms of a rating value (XMAC) as a simple percentage:

$$XMAC = PMAC$$

The least impacted system would be defined as having zero percent cover and the most impacted system as having 100 percent cover. Equivalent values of XMAC and PMAC are listed in Table 4.

A PROPOSED FORMULATION OF THE LEI

RATING VALUES AND TROPHIC LEVELS

The target variables used for evaluation are not mutually exclusive or independent variables and their comparison as a rating value for a given lake will not necessarily agree. Furthermore, each measures slightly different lake functions. Thus, the rating values are not expected to agree among themselves for a lake or set of lakes. This is apparent when comparing rating values for complementary variables such as macrophytes and chlorophyll a; however, the relationship between these variables, SD and the other target variables is sequential where one is a function of another.

The most difficult target variables to relate to problems or perceptions of problems are the nutrients nitrogen and phosphorus because they are causes, not effects. Also, nitrogen may be considered limiting on the basis of nutrient ratios (see USEPA, 1974; Miller, et al., 1974), but be supplied from atmospheric nitrogen by nitrogen fixing blue-green algae (Bartsch, 1972; Horne and Goldman, 1972; Schindler, 1977). Although ratios of nitrate to orthophosphate concentrations in lakes can be constant (Stumm and Morgan, 1970), the ratio does not allow interpretation of possible effects of nutrients directly. Loading and mass balance models (Vollenweider Approach) seem to offer the best approach to determining the effect of nutrient changes on lake quality and will be used to define trophic levels in relation to nutrient concentrations.

Morphological (depth) and hydrologic (flow through rate) factors affect significantly the nutrient, DO, macrophyte and chlorophyll a concentration in lakes. For these reasons it is important to look at the individual variables in terms of meeting the evaluation objectives, i.e., a comparison of the effects of specific lake restoration projects.

Various suggested levels of chlorophyll a concentrations have been related to trophic levels and Chapra and Tarapchak (1976) averaged these values to obtain reasonable quantitative definitions of trophic state (Table 6). Similar values have been estimated, for SD, inorganic nitrogen (TIN) and orthophosphate (TIP) as well as other parameters.

Good agreement with the values in Table 6 was obtained when the USEPA (1974) ranked 209 NES lakes and, by summing percentile rankings for 6 separate parameters, provided a breakpoint of 500 for oligotrophic lakes and 420 for mesotrophic lakes. These totals correspond to average percentile limits of 83.3 and 70.0 for eutrophy and oligotrophy, respectively. Values of parameters corresponding to these percentiles (Table 7) indicated very narrow ranges "defining oligotrophic and eutrophic" over the rather broad spread of actual concentrations shown in Table 5 for the rating value. The values in Table 4 were plotted and then the levels associated with different trophic states (Table 7) were noted for comparison (Figure 2). The variables that define different trophic states agree surprisingly well and in defining selective limits, show that rating values of less than 45 indicate oligotrophy and values greater than 50 indicate eutrophy.

TABLE 6. SOME ESTIMATES OF EUTROPHICATION LEVELS ASSOCIATED WITH SPECIFIC VARIABLES THAT MEASURE LAKE QUALITY.

	OLIGOTROPHIC	EUTROPHIC	REFERENCE
Variables in LEI			
Chlorophyll *a* photic zone mean, $mg \cdot m^{-3}$	<4	>10	NAS-NAE, 1972
	0.3-2.5	5-140	Sakamoto, 1966
	<4.3	>8.8	Dobson, *et al.*, 1974
	<1	>6	Carlson, 1977b
Overall average	<2.75	>8.7	Chapra and Tarapchak, 1976
peak photic zone, $mg \cdot m^{-3}$	<3	>20	Landner, 1976
Secchi Depth, m	>6	<3	Dobson, *et al.*, 1974
TIN, $mg \cdot m^{-3}$		>300	Sawyer, 1947
TIP, $mg \cdot m^{-3}$		>10	
Non-LEI variables			
PRIMARY PRODUCTIVITY-$^{14}CO_2$ uptake			
mean daily, mg cm^{-2} day^{-1}	30-100	300-3000	Rodhe, 1969
total annual g cm^{-2} day^{-1}	7-75	75-700	
mean daily, mg cm^{-2} day^{-1}	<200	>750	Landner, 1976
HYPOLIMNETIC DO DEFICIT			
rate, mg $O_2 \cdot m^{-2}$ day^{-1}	<250	>550	Mortimer, 1941
NUTRIENT LOADING (at mean depth, $\bar{z}$)			
TP, $g \cdot m^{-2}$ yr^{-1}		>0.3 (20 m) >0.8 (100 m)	Vollenweider, 1968
TN, $g \cdot m^{-2}$ yr^{-1}		>4 (20 m) >1 (100 m)	
ALGAE, $number \cdot ml^{-1}$	<2000	>15,000	Landner, 1976
BIOMASS, mg l^{-1}	<1	>10	
CELL VOLUME, $mm^3 \cdot l^{-1}$	<5	>30	
ROTIFERS, $number \cdot l^{-1}$	<10	>250	
MICROCRUSTACEA, $number \cdot l^{-1}$	<1	>25	
SPECIES DIVERSITY	low (high is mesotrophic)	low	

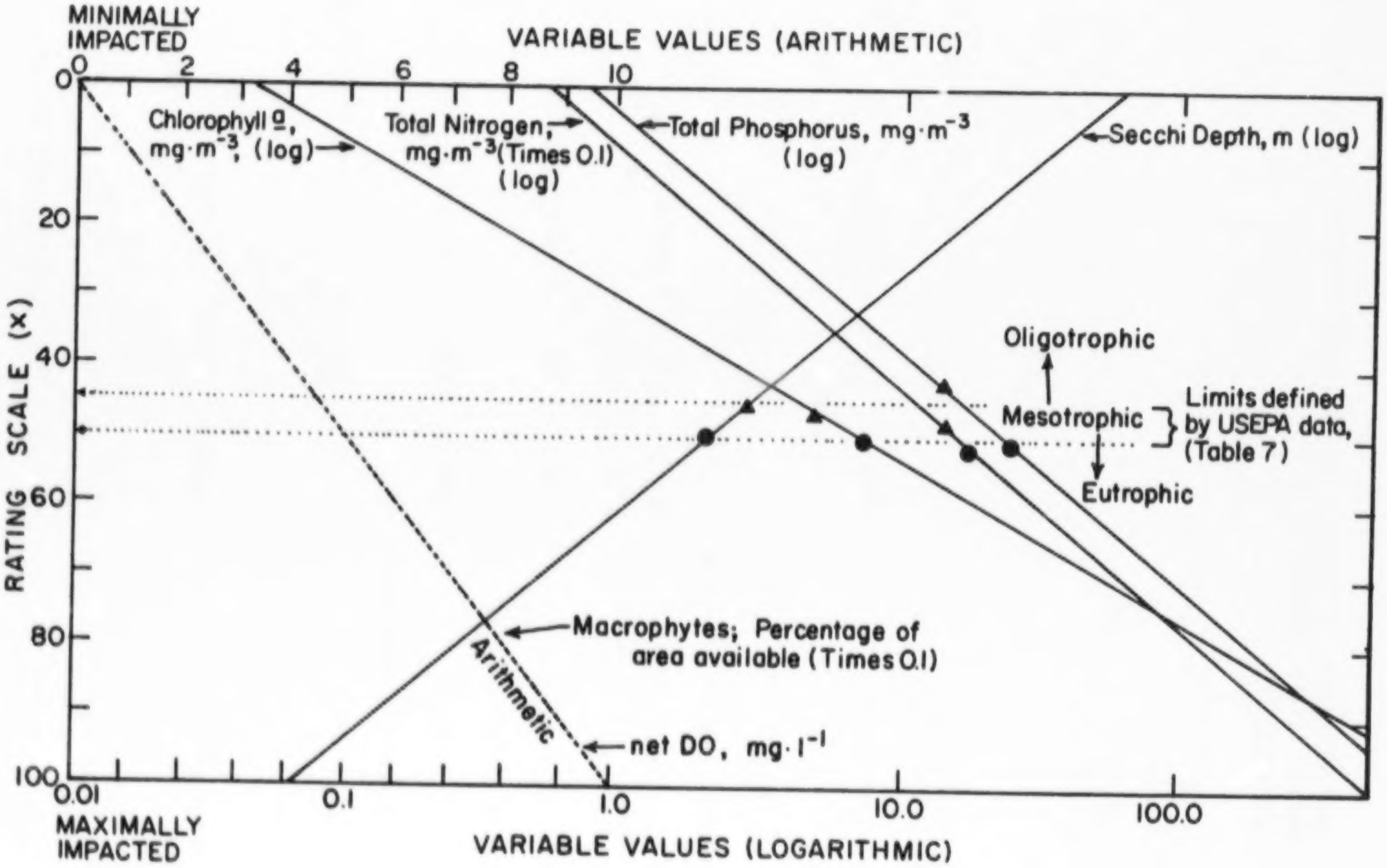


Figure 2. Relation between rating values and variable values compared to eutrophication levels.

TABLE 7. CONCENTRATIONS ASSOCIATED WITH TROPHIC STATE DEFINED BY A RELATIVE RANKING OBTAINED FROM NES DATA ON 209 LAKES (USEPA, 1974).

Parameter	Oligotrophic [percentile <83.3]	Eutrophic [percentile >70]
median total P, $mg \cdot m^{-3}$	<14	>25
median dissolved P, $mg \cdot m^{-3}$	<8	>11
median total N, $mg \cdot m^{-3}$	<140	>180
median chlorophyll a, $mg \cdot m^{-3}$	<4.8	>7.4
minimum observed DO, $mg \cdot l$	>7.2	<6.2
mean Secchi depth, m	>2.8	<2.0

* Estimation based on three sampling times (spring, summer, and fall) and 1 or more sampling sites and more than 1 sampling depth.

LEI

The formulation of the LEI was based on a number of assumptions, limited data, and as yet relatively untested concepts of the authors. The formulation of the LEI is hypothetical, and to a certain extent, arbitrary. It is proposed as an hypothesis that will be tested by applying synoptic data obtained from the evaluation grants or from literature data or NES data (USEPA, 1975b). The LEI is not intended to be unalterably structured. It is anticipated that testing the concept may result in some alteration of the LEI formulation.

The LEI has a range of 0 to 100 and was obtained by averaging specific target variables. Primary productivity in lakes is the sum of phytoplankton productivity and macrophyte productivity, therefore, the rating values of these two variables (XCA, XMAC) were summed and averaged; XSD and XDO were included directly; the nutrient variable was assumed to be XTP because of its typical importance but XTN could be (and will be for testing the hypothesis) substituted. Generally, if phosphorus is limiting, lower rating values for TN will be obtained than for TP. This comparison (XTP vs. XTN) is one way of determining whether to use XTP or XTN; i.e. the higher rating value of either XTP or XTN will be used. This resulted in the following equation:

$$LEI = 0.25\ [0.5\ (XCA + XMAC) + XDO + XSD + XTP]$$

As defined, the LEI is a simple number that ranges from 0 to 100 (minimal to maximal) and is related primarily to clean water uses as a function of eutrophication. Obviously, uses dependent on lake productivity such as fishing or largely unaffected by lake productivity such as irrigation storage are not related to the LEI. The formulation of such relationships will require utility functions. These utility functions would be cost/benefit functions, primarily but not exclusively. Utility functions for the LEI and lake use would include 1) optimality relationships (fishing, wildlife habitat), 2) linearly decreasing relationships (aesthetic, swimming, water supply, industrial uses), and 3) non-productivity affected relationships (irrigation, waste disposal, flood control, navigation). These would be influenced by the availability of such factors as alternative lake sites or water supplies and alternative activities or resources.

Also the LEI does not reflect other conditions such as toxicants (pesticides, heavy metals), salinity, inorganic sedimentation problems, spills. It is limited to productivity problems, i.e., eutrophication. Application to such problems would require modification and/or the development of other concepts.

SUMMARY

Two basic approaches, the Vollenweider or mass balance loading models and a lake evaluation index (LEI), are proposed to evaluate restoration manipulation(s) applied to a specific lake and to evaluate specific restoration techniques by studying a set of lakes. Although the Vollenweider Approach appears reasonable for phosphorus to a certain extent and has been accepted for managing lakes, a review of the literature reveals that little consensus exists on the development of indices for evaluating lake quality.

The LEI as proposed herein has a conceptual basis and includes the most commonly used target variables for limnological analysis of lakes: Secchi Depth (SD), Total Phosphorus (TP), Total Nitrogen (TN), Chlorophyll a (CA), net Dissolved Oxygen (net DO), and Percent Macrophytes (PMAC). Recommended sample collection and analysis appropriate to the LEI, the Vollenweider Approach, and associated necessary data are listed in Appendix A. Some of the target variables and associated data will be utilized in a quality assurance program to insure that comparable data are obtained.

Investigators on the evaluation grants will perform analyses appropriate for developing an understanding of lake limnology and the effects of restoration and analyses necessary to calculate the LEI and nutrient loadings. These will be used to assess treatment effects on individual lakes and to compare similar treatments on different lakes to achieve the objectives cited in the Introduction and to modify the LEI so that the most accurate and reproducible interpretation of lake response can be obtained. As the first step in this latter process a set of data for 28 lakes from the state of Washington have been evaluated using the LEI (Appendix C).

APPENDIX A

TABLE A-1. SAMPLING AND ANALYSIS FOR VOLLENWEIDER APPROACH

Variables	Min. Freq. of Sample Collection Season Spring	 Summer	 Fall-Winter	Time of Sampling	Sample Location	Data Source Or Analytical Method	References
I. Vollenweider Approach							
A. Water							
Precipitation	Continuous if Possible			Continuous	within lake margins	Class I weather station - nearest or most appropriate	USEPA, 1975a
Evaporation/Transpiration	"	"	"	"	" " "		
Inflow							
Tributaries	"	"	"	"	≥ 90% of inflow	USGS, irrigation, etc.	" "
Groundwater (±)	Accumulative or estimate			"	" " "	Lee, 1977	
Direct Runoff	Measure if possible or estimate			As Occurs	" " "	Wiers, precip/runoff, balance	" "
Outflow	Continuous if possible			Continuous	Outflow(s)	USGS, irrigation, etc.	" "
B. Nutrients							
Inflow Conc. (NO_2+NO_3-N, HN_4-N, TON, SRP, Total P)	Biweekly	Biweekly	Monthly	0900-1200	≥ 90% of inflow	See Table B-2 "	See Table B "
Inlake Conc. (same as inflow)	"	"	"	1000-1200	" " "	"	
II. LEI	"	"	"	1000-1200	deepest water at	"	"
A. Secchi depth	"	"	"	"	≥ 3 depths;	"	"
B. TP (see I.B.)					0.5 m, mid, bottom		
				"	" " "	"	"
C. TN "	"	"	"	"	" " "	"	"
D. CA	"	"	"				"
E. DO and Temperature Profiles	"	"	"	diurnal cycles if possible	profile at deepest point and in littoral zone	DO meter, thermistor	
F. Percent area covered by macrophytes		"~August 15"				survey. (Appendix C)	"
III. Associated and Optional Data							
A. Visual Observations - notes (weather, incident light (Weather stat, I.A.), climatic and lake conditions, any unusual situations, wildlife and human uses of lake and basin, air temp. lake evaluation)		each sampling event		1000-1200	Basin		
B. Morphometric: max. depth, depth area, depth volume curves, perimeter		once		-	within lake margins		
C. Surficial Sediments (< 15 cm)							
Bulk concentrations (Tot. N, Tot. P Avail. P, TOC, pH, Eh).		One period		any	" " "		
Release Rates (DO use, PO_4 release)		" "		Accumulative	" " "	SOD chambers	Sonzongni *et al*. 1977
D. Product. Data - C uptake, DO/CO_2 bottles		As appropriate		1000-1200	(as CA, II.D)		
E. 3 major algae and 2 major macrophytes		As in LEI		1000-1200	(as CA, MAC; II.D. & F.)		
F. Other variables		As appropriate		as approp.	within lake margins	As appropriate	
Pop. Surveys (macroinverts, fish zooplankton, bacteria)							
Other nutr. (inorgan. C. trace metal)							
Other cations and anions, organics, particulates (TDS, SS, VSS, etc.)							

APPENDIX B - MACROPHYTE EVALUATION

The following description has been adapted from the Wisconsin Department of Natural Resources guidelines for macrophyte surveys (Dunst, pers. comm.).

Evaluation of macrophytes as a target variable for use in the LEI.

Example (Pine Lake, WI) is included.

1. Obtain or draw a bathymetric graph in meters of the lake.

   Example. Figure B-1.

2. Divide lake surface into 100 sections (A/100) and draw squares with appropriate dimensions to form a grid over the lake surface. Label the North-South and the East-West axes numerically.

   Example. $\frac{6,070,000 \text{ m}^2}{100} = 60,700 \text{ m}^2\text{/quadrat}$

   $$\sqrt{60,700} = 246.4 \text{ m} \cong 250 \text{ m on a side.}$$

   NOTE: There are more than 100 squares defined; on Figure B-1, the grid pattern is indicated along the margins of the bathymetric map.

3. Define sections within the 10 m depth contour line. For this example all sections fall within the 10 m contour since the lake is shallower than 10 m. Select randomly 50 percent of these sections. For lakes smaller than 25 hectares, select a grid pattern which produces sections 50 m on a side; select randomly 50 percent of these sections.

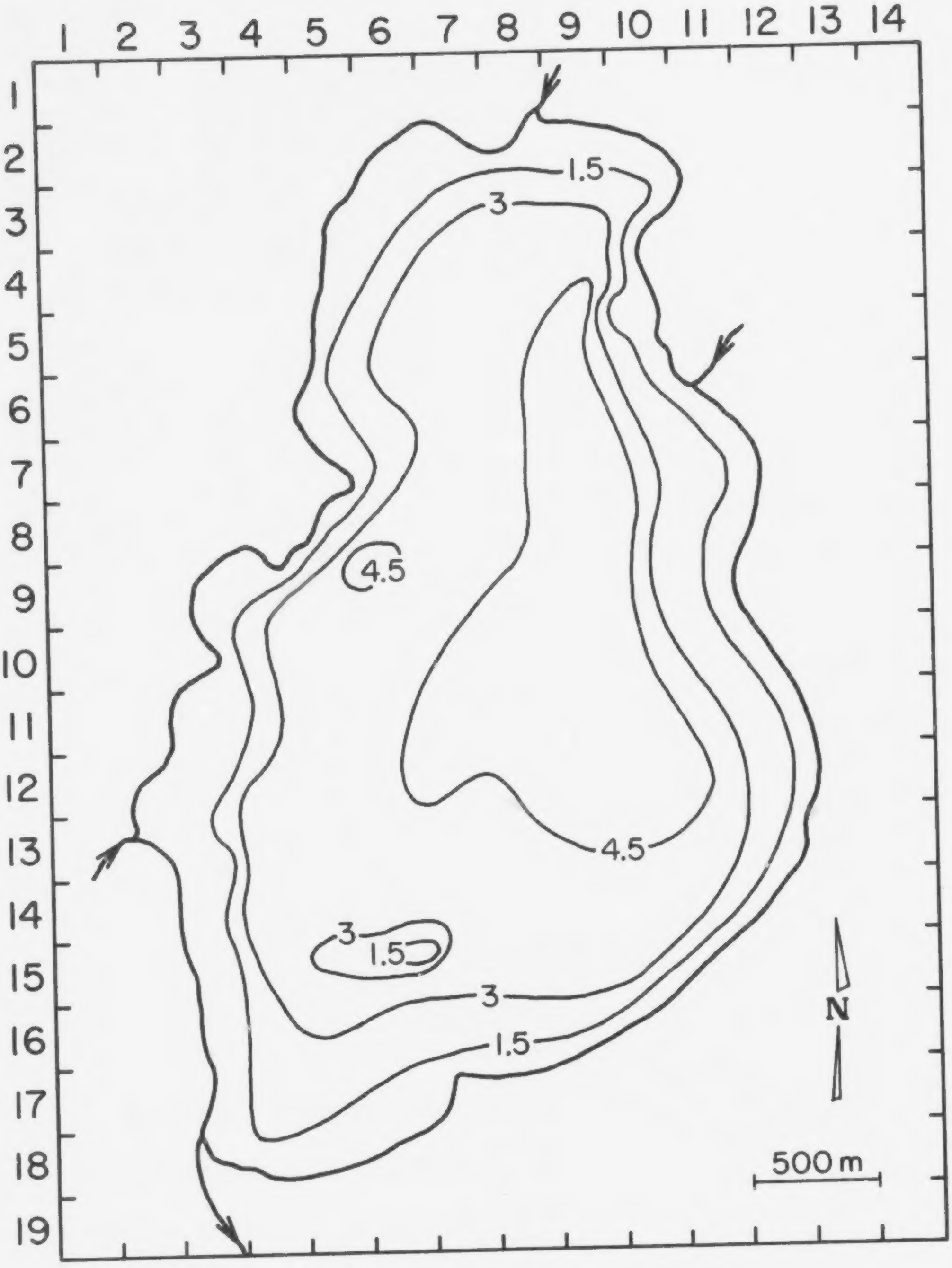
   Example. Figure B-1.

4. Mapping: Visually survey the lake and mark on the map the major community types: emergent, floating leaved and submergent plants.

   Example. Figure B-2.

   A = abundant
   C = common
   S = sparse

Indicate the boundaries of single species stands within the more general community type.


1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
1.5
3
4.5
4.5
3
1.5
3
1.5
N
500 m

PINE LAKE, WISCONSIN

Figure B-1. Bathymetric map of Pine Lake, Wisconsin, with grid pattern for selecting sections for macrophyte survey.

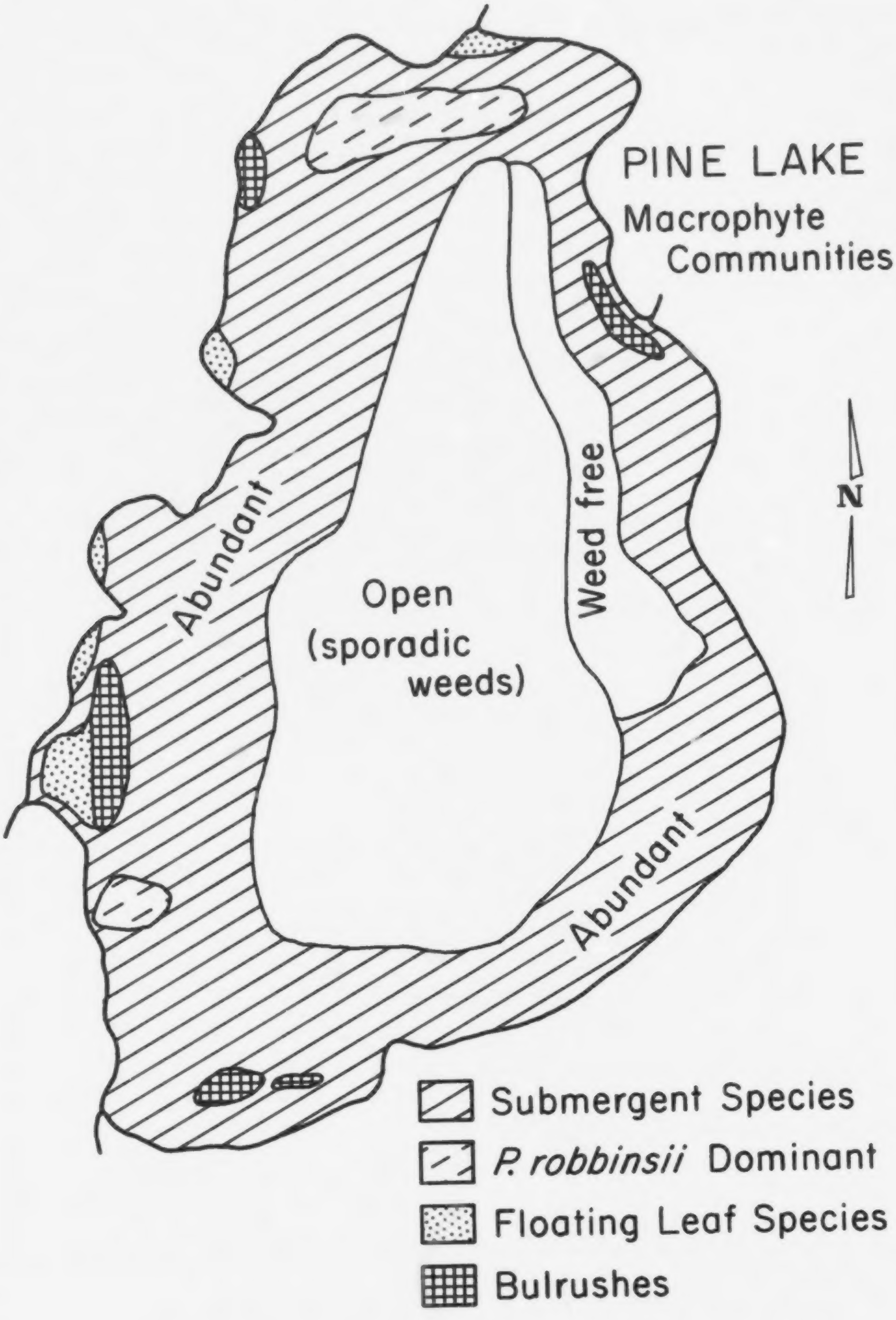

PINE LAKE
Macrophyte Communities
Abundant
Open (sporadic weeds)
Weed free
Abundant
N
Submergent Species
P. robbinsii Dominant
Floating Leaf Species
Bulrushes

Figure B-2. Map of Pine Lake, Wisconsin indicating distribution of major macrophyte types.

Show this information on a bathymetric lake map (Lind, 1974). The map should show distribution of the communities and a species list with the appropriate abundance symbol for each location (Fassett, 1960).

5. Density, frequency and depth. This analysis is applied to the selected sections as defined in steps 2 and 3.

Follow the grid pattern as much as possible using a compass and shoreline reference points, being reasonably precise.

Within the center of each selected section will be an imaginary circle with a six foot radius. Mentally divide the circle into quadrants and using an underwater viewer (Lind, 1974) determine the density of growth for each species according to:

1 = present in one quadrant
2 = present in two quadrants
3 = present in three quadrants
4 = present in four quadrants
5 = very abundant in all four quadrants

Visual determinations should be possible in most instances; however, a garden rake can be utilized if necessary to provide more reliable results. Additional measurements at each stations shall include:

a. Water depth (lake water level at the time of sampling should also be recorded),

b. percent of open surface area within the six foot radius,

c. sediment type (include combinations),

   i. rock
   ii. gravel
   iii. sand
   iv. muck--decomposed organic materials
   v. detritus--undecomposed organic materials (e.g., leaves, sticks, peat, etc.)
   vi. marl--whitish in color, fizzes profusely when muriatic acid is applied.

Reporting. In the report compute frequency occurrence, average density rating, and depth of growth for each species during each sampling period for the lake as a whole; however, furnish all of the original data. Numerically identify the approximate location of each sampling station on a map.

Indicate the total area available for macrophyte growth (10 m depth line and/or the contour line for 2 times mean Secchi depth for July-August), and the percent total lake surface covered by macrophytes.

APPENDIX C

APPLYING THE LEI (LAKE EVALUATION INDEX) TO WASHINGTON LAKES THAT ARE OF VARYING MORPHOLOGY AND TROPHIC LEVEL.

The LEI was developed for evaluating the effects of lake restoration techniques. Data adequate for illustrating the use of the LEI were obtained for a set of Washington lakes (Bortleson, et al., 1976). Four separate reports written by the Washington DOE and US Geological Survey on Washington lakes have been published; a single report was selected at random (Part 3) and all 28 lakes therein were evaluated. Data from the 28 lakes do not meet the specifications for the LEI described previously; they are reconnaisance data and were used only to illustrate the method for determining the LEI.

MORPHOMETRY (DEFINITIONS AS IN HUTCHINSON, 1957)

Areas (A), volume (V), perimeter (P), mean (ZB) and maximum (ZM) depth, and dissolved oxygen (DO) and temperature (T) relationships with depth were obtained along with Secchi depth (SD), total phosphorus (TP), total nitrogen (TN), chlorophyll a (CA), and percent area of the total lake covered by macrophytes. These data were used as input to a simple computer program for estimating the LEI. The development ratio (DL) was calculated as the ratio of the true perimeter to the circumference of a circle having the measured area of the lake. The diameter (D) of the lake can be determined assuming that the lake surface is a circle with area, A (Table C-1). The diameter can also be determined by assuming a geometric shape for the lake and by using the dimensions of this geometric shape to calculate D.

The calculation of various components of the LEI (particularly net DO) requires information about lake volumes which correspond to particular depth increments. If these volume increments are not available, they can be calculated by the method outlined below. The method assumes a lake to correspond to a particular geometric shape.

A cone shape has been suggested as a possibility (Hutchinson, 1957) but analysis of lake data indicated that a paraboloid would provide a better approximation of lake volume. The idealized relationships for conic and parabolic shapes compared to a hypothetical vertical plane of a lake can be visualized as in Figure C-1; certain types of morphometric configurations would produce ratios of the empirically determined area (Ae) to such volume-based areas (Av) that vary from unity depending on whether the actual plane was less or greater than the idealized planes shown in Figure C-1.

As a comparison of the goodness of fit by either the conic or parabolic shape, 1) the surface diameter (D) was calculated from the empirically determined volume using the conic (dvc) and then the parabolic (dvp) equations;

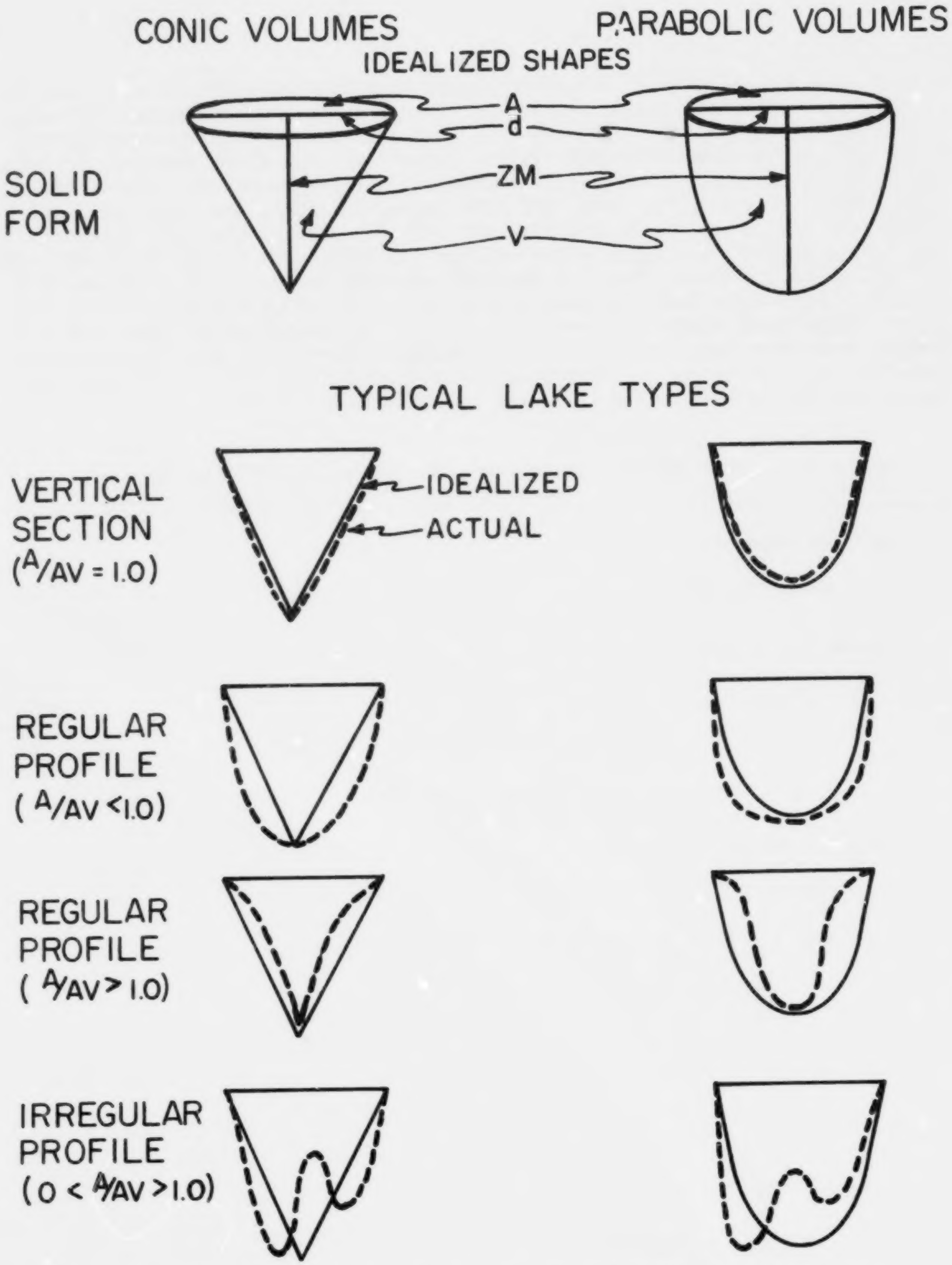

CONIC VOLUMES
PARABOLIC VOLUMES
IDEALIZED SHAPES
A
d
SOLID FORM
ZM
V
TYPICAL LAKE TYPES
VERTICAL SECTION ($A/AV = 1.0$)
IDEALIZED
ACTUAL
REGULAR PROFILE ($A/AV < 1.0$)
REGULAR PROFILE ($A/AV > 1.0$)
IRREGULAR PROFILE ($0 < A/AV > 1.0$)

Figure C-1. Use of geometric figures to approximate morphology of specific lake types (vertical dimension exaggerated).

2) the area was calculated from each diameter using the equation of a circle; and 3) these idealized areas were compared to the empirically determined areas using ratios. These ratios (Table C-2) indicate that the paraboloid approximates the set of 28 Washington lakes better than the conic because 1) more ratios of calculated to measured areas for the parabolas are closer to 1.0 (the ideal) than for the cone (20 were better; 3 about the same and 5 were worse); it should be noted that the cone did fit some of the lakes better; 2) based on a t-test the mean ratio of the 28 lakes for the parabola was not significantly different from 1.0 ($p<0.99$) whereas that for the cone was different. Also, note that the cone and parabola ratios were significantly different from each other. Attempts to correlate deviation of the idealized shapes from measured shapes with development ratios (DL) were unsuccessful. Essentially, the ratios of calculated to measured areas for the 28 lakes represent a normally distributed set of data with mean of 1.0 (Figure C-2).

TABLE C-1. MORPHOMETRIC ALGORITHMS FOR USE IN DETERMINING THE LEI.

1. Surface Area (A):

$$A = \pi r^2 = \frac{\pi d^2}{4}$$

2. Perimeter (P) and development ratio (DL):

$$\text{circumference} = \pi d$$

$$d = \sqrt{\frac{4A}{\pi}}$$

$$DL = P/\pi \sqrt{\frac{4A}{\pi}} = P \sqrt{4\pi A}$$

3. Lake volume (V) ratios

   a. Conic

$$V = \frac{1}{3} \pi r^2 h = \frac{1}{12} \pi d^2 h = \frac{1}{12} \pi d^2 ZM$$

$$d_v = \sqrt{\frac{12V}{\pi ZM}}$$

   b. Parabolic

$$V = \frac{\pi\ Y^2}{2a} = \frac{\pi\ ZM^2}{2a}$$

$$a = \frac{\pi\ ZM^2}{2V}$$

$$d_v = 2\sqrt{y/a} = \sqrt{8V/\pi ZM}$$

TABLE C-2. COMPARISON OF 28 WASHINGTON LAKES INDICATES THAT PARABOLOIDS APPROXIMATE LAKE VOLUME BETTER THAN CONES.

	Ratios	
	Conic area/ measured area	Parabolic area/ measured area
	1.590	1.060
	2.080	1.387
	1.396	.931
	1.402	.934
	1.201	.801
	1.200	.800
	1.380	.920
	1.766	1.178
	1.544	1.029
	.969	.646
	1.378	.918
	1.600	1.067
	2.052	1.368
	1.799	1.199
	1.921	1.281
	1.119	.746
	1.380	.920
	.943	.629
	1.309	.873
	1.091	.727
	1.199	.799
	1.024	.682
	1.219	.813
	1.255	.837
	1.722	1.148
	1.777	1.185
	1.446	.964
	1.898	1.266
mean	1.45	0.968
Standard deviation	0.326	0.217
Standard error	0.062	0.041
Range	1.137	0.758
t-value (compare to 1.0)	7.35*	0.78**
t-value (compare conic to parabolic) = 6.54*		

* significantly different at P > 0.99

** not significantly different at P < 0.95

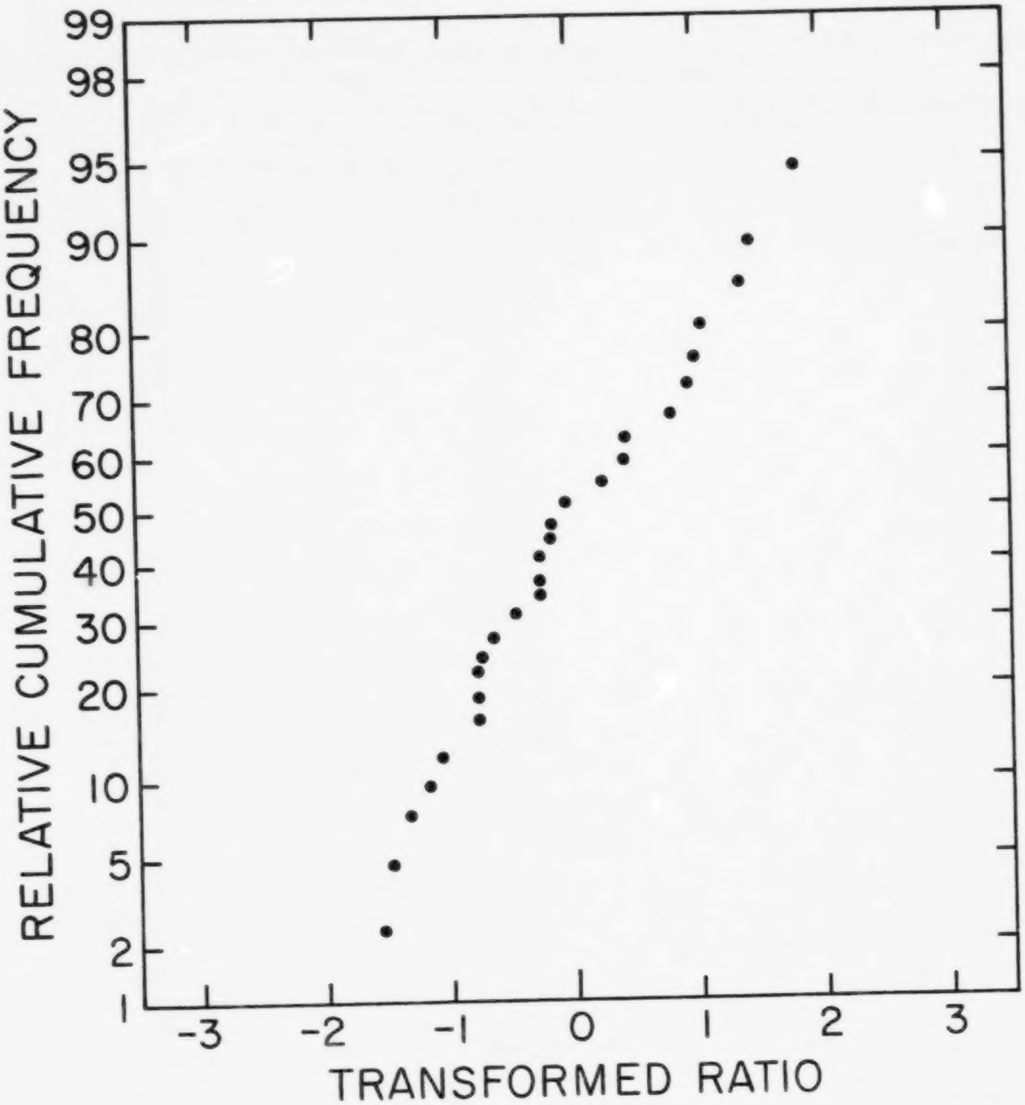


Figure C-2. Relative cumulative frequency of ratios of surface areas calculated (parabolic) to empirical surface areas plotted on probability graph paper. Horizontal axis was obtained from

$$u = \frac{y-\mu}{\sigma}$$

where y = observation, μ = mean of observations and σ = standard deviation. This normalizes the deviations to the standard deviation producing u = 0 for the mean, and each unit on the axis = one standard deviation. A normal distribution plots as a straight line on this graph.

Translating idealized volumes from either cones or paraboloids is simplified by certain definitions (Figure C-3): depth is the positive portion of the axis in the x, y plane; the plane is rotated around the axis to produce the solid figure in the x, y, z dimensions; radius (d/2) is a ±x value for the plane.

Specifically, volume increments (V_i) are calculated as follows: because depth (D) is usually measured from the surface, H = ZM-D. Similar triangles are formed in the cone ($\alpha\beta\gamma$ similar to $\sigma e \gamma$) and thus the ratio of any two sides is a constant ($k = d/ZM = \sigma e/Y$). The diameter (d_i) at any depth (D) would be: $d_i = k(ZM-D)$. Thus, the volume of a cone (Vc) contained below any specified depth (D) can be calculated:

$$V_c = \frac{\pi}{12}[k(ZM-D)]^2(ZM-D) = \frac{\pi}{12}k^2(ZM-D)^3$$

or because $k = \sqrt{\frac{4A}{\pi ZM^2}}$ and $H = ZM-D$

$$V_c = \frac{\pi}{12}\frac{4A}{\pi ZM^2}H^3 = \frac{AH^3}{3ZM^2}$$

The incremental volume (V_i) is the total volume minus the conic volume:

$$V_i = V - V_c$$

The area for that conic volume is easily calculated assuming a circle for the base of the cone, V_c.

PARABOLIC

The equation describing a parabola with the spatial orientation in Figure C-3 has the form $Y = ax^2$. A parabola rotated around the axis (Y) produces a paraboloid having a volume (V_p):

$$V_p = \frac{\pi Y^2}{2a}$$

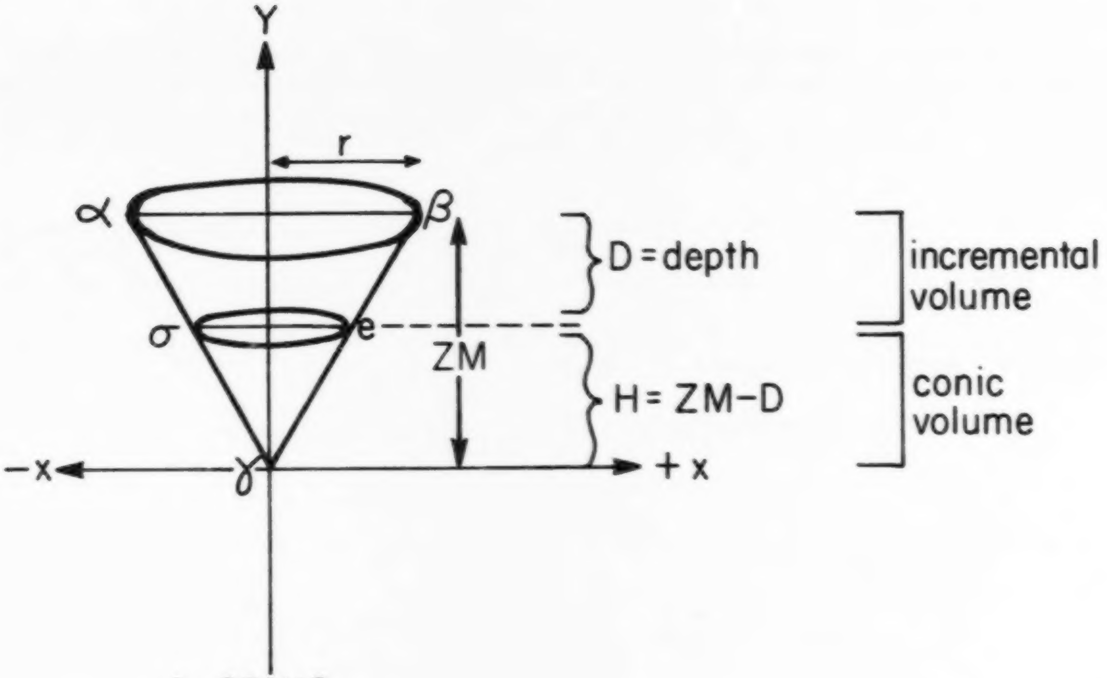
The coefficient, a, is defined for a lake having maximum depth (ZM) and known volume (V) as follows:

$$a = \frac{\pi ZM^2}{2V}$$

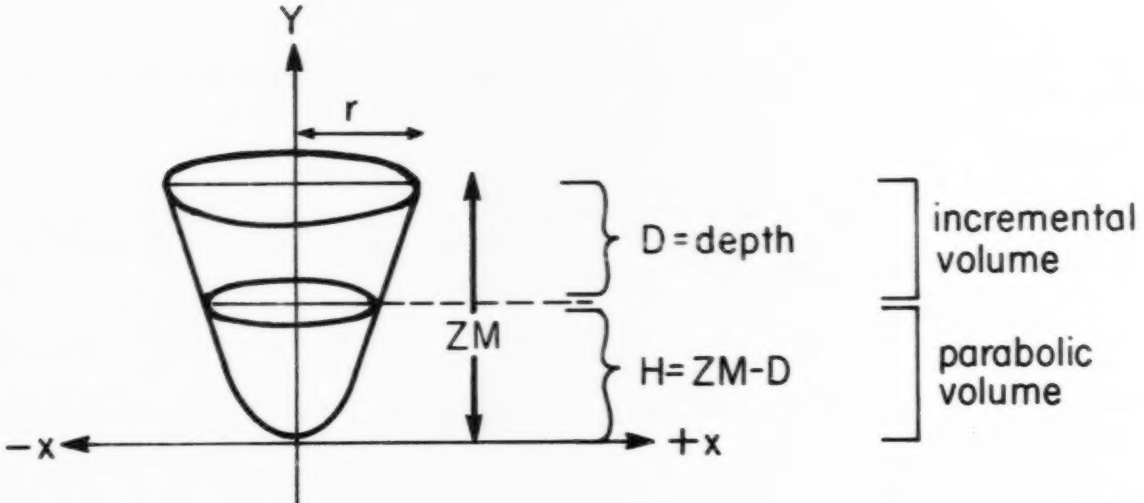
Thus, $V_p = \frac{VY^2}{ZM^2}$

Incremental volumes (V_i) for any depth (D) where Y = H = ZM-D are:

$$V_i = V - V_p = V - \frac{VH^2}{ZM^2} = V\left(1 - \frac{H^2}{ZM^2}\right)$$


Y
r
α
β
D = depth
incremental volume
σ
e
ZM
H = ZM−D
conic volume
−x
+x
γ

A. CONIC


Y
r
D = depth
incremental volume
ZM
H = ZM−D
parabolic volume
−x
+x

B. PARABOLIC

Figure C-3. Spatial definitions used to compute idealized volumes of lakes with conic or parabolic basin configurations.

The area for that paraboloid, V_p, is determined from the area of a circle for the base of the parabola (Table C-2).

EXAMPLE

Goodwin Lake, WA is shown in Figure C-4 along with DO and temperature profiles with depth for spring and summer, 1972. Interpolated data for 2 dates for oxygen and temperatures at specific depths are shown in Table C-3. Table C-4 summarizes the data and displays the transformed values calculated with the equations described previously. Although the example data are sparse and do not rigidly meet the requirements listed in the text, they illustrate the process.

LEI values were calculated for each of the 28 Washington lakes mentioned previously, and the lakes were ranked in ascending order. This ranking was then compared with qualitative describers given in Bortlesen *et al*. (1976). Table C-5 demonstrates relatively good agreement between the LEI values and the qualitative describers of trophic character; the low LEI values tend to correspond with lakes of low biological productivity and the higher value with lakes of higher biological productivity.

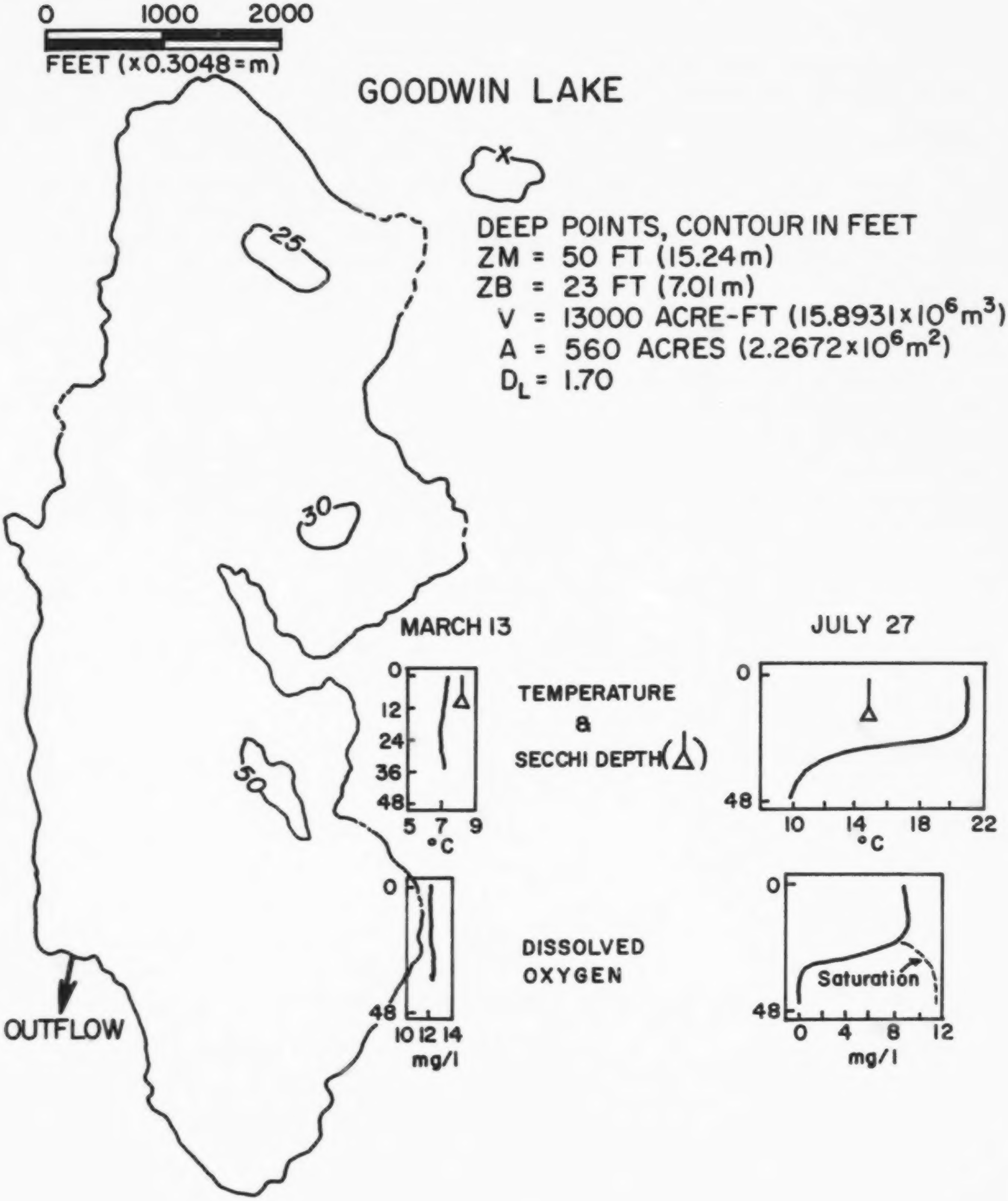

0 1000 2000
FEET (x0.3048=m)
GOODWIN LAKE
X
DEEP POINTS, CONTOUR IN FEET
ZM = 50 FT (15.24m)
ZB = 23 FT (7.01m)
V = 13000 ACRE-FT (15.8931x10⁶m³)
A = 560 ACRES (2.2672x10⁶m²)
D_L = 1.70
25
30
50
OUTFLOW
MARCH 13
JULY 27
TEMPERATURE & SECCHI DEPTH
0 12 24 36 48
5 7 9
°C
10 14 18 22
DISSOLVED OXYGEN
10 12 14
mg/l
Saturation
0 4 8 12

Figure C-4. Areal dissolved oxygen deficit calculations are determined based on above data traced from Bortleson, et al. (1976), for Goodwin Lake as an example of 28 other lakes taken from the report.

TABLE C-3. DATA ON GOODWIN LAKE FROM BORTLESON et al., 1976 (See Figure C-3).

Depth (m)	Spring (March 13) DO (mg·l⁻¹)	Spring (March 13) Temp. (°C)	Summer (July 27) DO (mg·l⁻¹)	Summer (July 27) Temp. (°C)
0 (surface)	12.	7.	8.8	21.
6.4	unchanged		8.8	21.
7.3	to bottom		8.8	19.
9.8			0	12.
13.4			0	10
15.24 (bottom)			0	10

TABLE C-4. DATA NEEDED FOR LEI USING EXAMPLE OF GOODWIN LAKE (FROM BORTLESON et al., 1976).

Variable	7/27/72 Data*	Calculated LEI data (variable)
Secchi Depth (m)	4.27	39 (XSD)
Total P (μg·l⁻¹)	12.	73 (XTP)
Total N (μg·l⁻¹)	820.	73 (XTN)
Chlorophyll a, (μg·l⁻¹)	12.3	55 (XCA)
Net Dissolved Oxygen (mg·l⁻¹) [Measured-Saturated from Table C-3]**	2.52	25 (XDO)
Percent total area covered by macrophytes	1.0	1.7 (XPMAC)
LEI		41

* Transform of only one data point; in practice the average of weekly data collected in July and August would be used. For DO, the average of the calculated total differences would be used.

** These data corrected for temperature:

$$\text{DOSAT} = 522/(36 + 0.5T).$$

If lake is not at sea level, correction for pressure should be made

$$\left(f = \frac{\text{actual P}}{\text{760 mm P}}\right)$$

TABLE C-5. COMPARISON OF LEI VALUES FOR 28 WASHINGTON LAKES WITH ESTIMATED TROPHIC STATE (BIOLOGICAL PRODUCTIVITY, BORTLESON *et al*. 1976).

Lake	LEI	Trophic Character (biological productivity)
Wye	27.48	Low
Phillips	29.70	Low
Retreat	31.36	Medium
Goodwin	33.19	Medium
Wallace	33.79	Low
Ward	34.93	Low
Mason	35.10	Low
Walker	35.87	Low to Medium
Offutt	36.64	Medium High
Roesiger (North Arm)	36.97	Low to Medium
Mineral	37.23	Moderate
Echo	37.35	Medium
Stevens	39.88	Medium
Roesiger (South Arm)	40.96	Medium
King	44.25	Medium
Deer	44.92	Low to Medium
St. Clair (North Arm)	47.82	Moderate to High
Diamond	47.84	Medium
Hicks	48.16	Moderately High
Heritage	48.57	High
Boren	49.34	Medium to High
Pierre	50.80	Moderate to High
St. Clair (South Arm)	52.45	Moderate to High
Thomas	53.42	Medium
Leo	53.65	Medium to High
Frater	55.79	Medium to High
Sherry	55.81	Medium
Gillette	63.06	Medium to High

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EVALUATION OF CLEAN LAKES RESTORATION USING PHOSPHORUS MASS BALANCE MODELING

by

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BACKGROUND

One way of assessing the response of lakes to remedial treatment is to measure how selected variables change after treatment and perhaps to combine these variables in a way which can be used to compare how well lakes respond to similar or different treatments relative to one another. This comparative assessment is one of the purposes for developing the Lake Evaluation Index (LEI: Porcella and Peterson, 1977). The design of the LEI does not incorporate the dynamic connection between treatment and response; that is, for a given level of treatment, what kind of change can be expected in selected lake properties? The following describes how we propose to use current chlorophyll a (chla) - phosphorus (P) modeling techniques to assist in the evaluation and prediction of the response of lakes subject to remedial treatment.

There has accumulated a remarkable amount of data over the past several years which demonstrates a high correlation between the concentration of algae (as measured by chla) in lakes and the P content of P limited lakes (see review in Nicholls and Dillon, in press). The relationships cover broad ranges of both chla and P, generally several orders of magnitude. But, although this high degree of correlation occurs over the broad range, at any particular level of P, high variablility occurs in the amount of chla. For example, Dillon and Rigler (1974a) related summer chla (mid-May to mid-September) in the euphotic zone and springtime total P as:

$$\mathrm{Log}_{10}\,[\mathrm{chl}\underline{a}] = 1.449\,\log_{10}\,[P] - 1.136 \qquad (1)$$

Ninety percent of the variation in chla could be accounted for by variation in springtime total P for the lakes examined. However, the variance in chla at any particular P concentration was high. For example, when P = 20 µg/l, the expected chla is 5.61 µg/l with 95% confidence limits of 2.06 - 15.3 µg/l.

Nicholls and Dillon (in press) suggest that a closer correspondence might exist if algal volume rather than chla were used as the measure of algal biomass. Because insufficient data are presently available to determine the generality of their assertion, we will rely mainly on chla-P relationships, but suggest the importance of obtaining phytoplankton volume estimates.

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We attempt to predict the P content of lakes because of the agreement between chl*a* and P content and because we can analyze the consequences of changing P supplies to lakes which occur as a result of changing management practices in watersheds. Since the P content of a lake occurs as a balance between supplies and losses (both external and internal), accurate measurement of all fluxes from sources and to sinks would provide an accurate description of changes in lake P. Unfortunately, it is difficult (if not presently impossible) to measure all these fluxes accurately, particularly those relating to internal sources and sinks. Therefore some indirect methods must be used.

A mass balance description of P in lakes can be written:

$$\text{change in lake content} = \text{external supply} - \text{outflow} + \text{internal supply} - \text{sedimentation}$$

$$V\,\frac{d[P]}{dt} = J_{ext} - Q\,[P] + J_{int} - S \qquad (2)$$

where $[P]$ = lake P concentration (mg/m^3)

J_{ext}, J_{int} = external and internal supply rates (mg/wk)

V = lake volume (m^3)

Q = water outflow rate (m^3/wk)

and S = sedimentation rate (mg/wk)

The lake P content and the external P supplies and losses through the outlet can generally be measured with reasonable accuracy. More difficult to measure are the internal source-sink terms. However, by examining how much of the change in P in lakes can be accounted for by differences between external supplies and losses, the importance of the internal source-sink term can be evaluated. This idea has been used to determine how much inflowing P deposits annually and to determine relationships between the fraction of inflowing P which deposits and hydrologic and morphometric properties of lakes (Dillon and Rigler, 1974b; Chapra, 1975; Dillon and Kirchner, 1976; Larsen and Mercier, 1976; Vollenweider, 1976).

These relationships along with P loading information can be used to predict the average P content of lakes given the necessary morphometric and hydrologic data. The equations derive from equation (2) for certain assumptions: 1) internal P loading is insignificant, 2) P deposition occurs as a constant fraction of the P present, 3) lake is well mixed, and 4) water inflow and outflow are steady and equal and lake volume is constant (see Dillon and Rigler, 1974b; Vollenweider, 1975; Sonzogni, Uttormark, and Lee, 1976). The equations are:

$$\frac{d[P]}{dt} = \frac{J_{ext}}{V} - \rho_w\,[P] - \sigma_\rho\,[P] \qquad (3)$$

The time dependent solution is:

$$[P] = \frac{J_{ext}}{Q}\left(\frac{\rho_w}{\sigma_\rho + \rho_w}\right) - \left[\frac{J_{ext}}{Q}\left(\frac{\rho_w}{\sigma_\rho + \rho_w}\right) - [Po]\right] e^{-(\rho_w + \sigma_\rho)t} \qquad (4)$$

At steady state:

$$[P] = \frac{J_{ext}}{Q}\left(\frac{\rho_w}{\sigma_\rho + \rho_w}\right) \qquad (5)$$

where ρ_w = annual water washout coefficient = $\frac{Q}{V}$ (yr^{-1})

σ_ρ = annual fractional loss of lake P to sediments (yr^{-1}).

Since a direct determination of σ_ρ is difficult, alternate methods for estimating its value have been suggested. Dillon and Rigler (1974b) showed that σ_ρ could be estimated by determining how much inflowing P left a lake through the outlet for a lake in a steady state. For lakes in which the P content changes over the year, the net annual flux of P to the sediments can be estimated from input-output differences if the change in lake P is included. Dillon and Rigler (1974b) showed that $\sigma_\rho = R\,\rho_w/(1-R)$, where R was the fraction of inflowing P which sediments annually. Equations 4 and 5 can be rewritten as:

$$[P] = \frac{J_{ext}}{Q}(1 - R) - \left[\frac{J_{ext}}{Q}(1 - R) - [Po]\right] e^{-\left(\frac{\rho_w}{1 - R}\right)t} \qquad (6)$$

$$\text{and } [P] = \frac{J_{ext}}{Q}(1 - R) \qquad (7)$$

R can be determined by measurement (Dillon and Rigler, 1974b) or by using one of the empirical (Larsen and Mercier, 1976; Vollenweider, 1976) or theoretical (Chapra, 1975) expressions. These expressions are:

$$R = \frac{1}{1 + \sqrt{\rho_w}} \qquad \text{(Larsen and Mercier, 1976)}$$

$$\text{and } R = \frac{v}{v + q_s} \qquad \text{(Chapra, 1975)}$$

where v = net annual settling velocity (m/yr) and

q_s = areal water loading = Q/lake surface area.

Equations 6 and 7 form the basis for projecting expected changes in phosphorus when P input supplies are altered.

In many lakes, although net deposition occurs on an annual basis, the rate of deposition might not be constant (or a constant fraction of the amount of phosphorus in the lake), or net release from the sediments might occur over periods of up to months (for examples, see Ahlgren, 1976; Welch, 1977; Larsen et al., 1975). These events, especially sediment P release, might be important in controlling algal productivity and bloom formation particularly during summer months. To identify these short term events, equation (2) can be rewritten to form the analytical framework:

$$J_{int} - S = V \frac{d[P]}{dt} - J_{ext} + Q[P] \qquad (8)$$

By measuring all the terms on the right side of equation (8) precisely on a weekly or biweekly frequency, the net source-sink flux can be determined by difference. Essentially, the change in lake P content (V d[P]/dt) not attributable to differences between surface supplies and losses is attributable to the internal source-sink term.

It must be emphasized that this method estimates the net flux of P to or from the sediments. Conceptually it is similar to the method described by Dillon and Rigler (1974b) for experimentally determining the annual retention of P in lake sediments, but equation 8 uses shorter time intervals to determine when major internal fluxes occur. Measurements of gross fluxes of P to or from sediments requires a considerably more elaborate experimental program. This program might include the measurement of P release by isolating sections of the lake bottom with cores incubated in the lab (Bannerman et al., 1974), or by using submerged chambers (Sonzogni et al., 1976; Welch, 1977); measurement of deposition with settling traps (Håkanson, 1976; Kimmel et al., 1977); and assessment of the importance of macrophyte communities as a source/sink of P (Lie, 1977). Since many of the techniques for measuring gross flux are not yet standard and might require resources which cannot presently be supplied by Clean Lakes funding, it is probably infeasible to experimentally determine gross fluxes at this time.

An evaluation of the timing and magnitude of net internal fluxes can be used to identify factors which control these fluxes. For example, in shallow lakes, high winds might resuspend sedimented P. When quiscent conditions are re-established, algal blooms might develop using the store of dissolved P which was stirred up. Or sediment release of P and accumulation in the hypolimnion can occur when anaerobic conditions develop in large lakes. Then, erosion of the thermocline can transfer significant amounts of P into the epilimnion with the potential of stimulating algal proliferation (Stauffer and Lee, 1973). Careful measurements, using equation 8 as a framework, can show the occurrence of these events. Correlation with other environmental variables can lead to an establishment of causal factors.

An assessment of internal fluxes can also be used evaluate the effectiveness of restorative treatments directed at reducing sediment P supply. Nutrient inactivation and dredging are two techniques which attempt to reduce that supply. A before-after evaluation can show how well the internal fluxes have been altered by treatment. Cooke et al. (in press) used basically this technique in analyzing the effectiveness of hypolimnetic application of alum in Twin Lakes, Ohio. Their results suggested that hypolimnetic application of alum was ineffective in reducing epilimnetic P concentrations significantly because most of the epilimnetic P was probably supplied from epilimnetic sediments.

OBJECTIVES

This portion of the evaluation of the response of Clean Lakes to remedial treatment will use equations (1)-(8) as the basis for projecting the lake's expected response and evaluate the effectiveness of remedial treatment.

Specifically, we propose to:

1) Quantify the changes in external and internal P supplies to selected lakes resulting from remedial treatment and project the expected changes in lake P content resulting from reduction in P supplies.

2) Quantify the seasonal changes in P content of selected lakes to determine the importance of internal supplies of P and to evaluate the effectiveness of treatment methods designed to reduce internal P loading.

3) Relate the development of algal biomass to the P content of selected lakes and to show how algal biomass is expected to change with reductions in lake P content.

4) Determine how well the Lake Evaluation Index (Porcella and Petersen, 1977) relates to the P content of lakes.

Data Base and Suggested Frequencies for Sampling

The following list of environmental variables and sampling frequencies are offered as guidelines, and are not meant to make up a rigid protocol. Certainly each principal investigator understands the system which he is assessing best, so he should tailor a sampling program to optimally quantify the treatment and the effect within the lake of interest.

1. Water budgets: Obtain the best estimates of flows from tributaries, rainfall, overland flow, storm water inflow and lake outflow. Sampling should be structured to obtain measurements of the most important contributors most frequently, particularly during storm events. Also sampling frequency should be higher for sources contributing the most P.

2. Total phosphorus (TP), total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP): Sample above identified sources. Frequency of sampling should be dictated by variability of P concentrations as well as magnitude of particular sources. Those sources of greatest impact should be sampled weekly, or more often if possible; use of flow-weighting

compositors is recommended for the major sources. Events which are likely to cause large inputs of P over short periods of time should be sampled intensively. Flow measurements should accompany P sampling.

3. In-lake TP, TDP, SRP: Sample vertical profiles at one or more sites in the lakes, the number of sites depending on the spatial variability in the lake. Depth frequencies should be adequate to describe the profile (1.5-2m if stratified; less frequently if well mixed). A suggested frequency is biweekly or weekly (especially if temporal variability is severe). Consideration should be given to augmenting vertical profiles with volume-weighted samples obtained at other sites in the lake. Sample intensively when windstorms or cold fronts are likely to generate turbulence which might redistribute P within the lake or resuspend.

4. In-lake chlorophyll *a* (corrected for phaeophytin)--sample in a manner similar to P sampling. Note: Regarding the standard acetone methanol extraction procedures for extracting chl*a*, recent experiments using an acetone-DMSO (dimethyl sulfoxide) mixture have improved extraction efficiencies for algae for which complete chl*a* extraction has been difficult (see Shoaf and Lium, 1976; Stauffer and Armstrong, 1977). Some of the variation in the chl*a*-total P relationships might be attributable to incomplete extraction of chl*a* when certain algae (blue-greens, greens) become dominant.

5. In-lake temperature--vertical profiles with P sampling, particularly to define thermocline.

6. In-lake dissolved oxygen, alkalinity, conductivity, nitrogen compounds: Sample at least 1 station, vertical profile, frequency as for P.

7. Lake morphometry--surface area and volume hypsograph: Obtain seasonal variation in lake volume (lake level) if such variation is significant.

8. Meteorology: Windspeed and direction (daily)
   Air temperatures (min - max)(daily).
   Solar radiation (daily).
   Light extinction (weekly).

9. Macrophyte community--evaluate likely interaction between macrophyte areas and open lake water from perspective of phosphorus flux. Estimate areal distribution at maximum density.

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SOCIAL EVALUATION OF THE CLEAN LAKES PROGRAM

by

Neils Christiansen*

INTRODUCTION

This paper describes the research which should be completed in the socioeconomic evaluation of the Clean Lakes Program. It begins with a brief statement of the rationale for such research and of the information needed by decision makers who are considering whether or not to undertake a lake restoration. Next is a discussion of some of the factors which serve to set the scope of the research to be initiated during the next year or so. Finally, the objectives which should be fulfilled as part of the research currently underway, or to be initiated during the coming year, are set forth along with some description of the issues which need to be considered in meeting those objectives.

The next step will be to evaluate the particular activities of each of the current grantees using this statement as a guide. The result will be a specification of the work still in need of undertaking by any future grantees.

RESEARCH PHILOSOPHY

Social research in the Clean Lakes Program is interested in the entire process of human concern, action and consequences of lake restoration and protection. The research is therefore concerned with the kinds of questions commonly encountered by all the social science and managerial disciplines. The researcher views a restoration effort as one point on a time continuum of stimulus and response which began with man's first involvement with the lake, and which will continue forward more or less indefinitely.

The questions of interest to the social researchers thus include the following: What forces from outside the current community have affected man's involvement? What sort of attitudes exist with regard to the role of the lake in individual and community affairs? What process led to the current concern over the condition of the lake? What is the power structure in the community and how has that structure affected thinking about the lake? Who will benefit from lake restoration? What form will the benefits take? Who will bear the burden of the costs and what form will the costs take? What are the relative importance of the different benefits and costs, i.e., how will the community trade them off one against another?

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In attempting to answer the questions above, the social research in the Clean Lakes Program should provide information which will be helpful in making decisions about possible future lake restorations or protection. The decision makers in need of such information may be the general citizenry, private firms or public officials at the local, state, or national levels. The general nature of the necessary information is clear: what are the resources required, the benefits derived, and how does the anticipated net benefit compare with the alternative uses for the resources? Thus, in essence, what is needed are the elements of a benefit-cost analysis. However, to be appropriate, the benefit-cost analysis must be viewed in broader terms than has sometimes been the case.

The end result of a benefit-cost analysis is sometimes considered to be a benefit-cost ratio.[1] The project is then considered to be justified or not, depending on whether the ratio is greater than, or less than, unity. Conceptually, such an approach is correct. Only those projects in which anticipated aggregate benefits are in excess of all costs are desirable. However, in practice, several conditions are usually present which make a benefit cost ratio an insufficient basis for decision making:

1. The benefits and costs may be unequally distributed among the population.

2. Some benefits or costs may not be measurable in financial terms, at least with acceptable precision.

3. Some significant benefits or costs may not have been recognized in the analysis at all.

The social evaluation of environmental issues to date indicates that all three of the conditions listed above may be present in any attempt to derive a benefit-cost ratio for a lake restoration effort. Therefore, in considering the merits of a project it seems likely that something is needed besides the traditional benefit-cost ratio.

In view of this fact, one might wonder why bother with a benefit-cost ratio at all. Why not concentrate on identifying as many of the social impacts as possible, then present a description of each of those impacts to the decision makers. Although such an approach is possible the results will often be very voluminous. This coupled with the variety of impacts (both positive and negative) may cause such an approach to be of little help to a decision maker. He may simply lack the time or ability to review and synthesize all the material. The combination of multiple and diverse impacts described in detail is often apparent in environmental impact statements.

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[1] Substituting the alternative measures of discounted (present) net worth, internal rate of return and equal annual equivalent does not alter the ideas which follow, although the standard of comparison will differ: zero for discounted net worth and equal annual equivalent and the guiding rate of interest for internal rate of return.

Such an approach has been characterized as "write down everything we know about the problem and hope it makes sense."[2] A better approach is to summarize, to condense the information to the essence. It is for just such reasons that benefit-cost analysis was developed in the first place, since many human values can be validly accounted for in financial terms. Therefore, what is needed is an analysis which identifies all the significant social impacts. These impacts should then be analyzed in an effort to state them in common terms so that trade-offs can be recognized with as little difficulty as possible. This process will, in most cases, lead to a final statement which contains both a financial analysis (such as a benefit-cost ratio) and some considerable description about the benefits and costs that are not adequately represented in the financial analysis.

INFORMATION NEEDS

The information needed by decision makers to arrive at an informed decision about whether or not to undertake restoration of a lake, what procedures to use and how intensively to use them requires, as a minimum, the four types of information listed below.

1. Water Quality

Information about the changes in water quality parameters which would result from treatment activities of various sorts. Ideally this information would be in the form of predicting equations, with one equation for each combination of water quality parameter, type of treatment and treatment technology. For example:

$$P_a = f\,(A,\ P_b, \ldots) \tag{1}$$

where P = phosphorus content before (P_b) and after (P_a) treatment

A = quantity of alum used

$$P_a = f(V_s,\ P_b, \ldots) \tag{2}$$

where V_s = the volume of sediment removed

P = as above

and where, in each case, there is a certain technology used.

Presumably the equations would be dependent on a number of other variables such as the area and depth of the lake. Also, new imputs to the lake would tend to offset the treatment so that strictly speaking, (1) and (2) are dynamic.

$$P_t = f(A_o,\ P_o, \Delta P/\Delta t,\ t, \ldots) \tag{3}$$

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[2] Inhaber, Herbert. 1975. Environmental Indices. John Wiley and Sons. New York. 178 pages. See p. 152.

2. User Demand Functions

Information which can be used to predict the number of users of different sorts (swimmers, fishermen, etc.) using a lake as a function of water quality parameters, other lake characteristics and the socioeconomic circumstances of the users. For example:

$$Q_s = f(B,C,S,Y,T,\ldots) \qquad (4)$$

where Q_s = number of swimming recreationists using the site

B = some measure of blue-green algae concentrations

C = clarity

S = surface area of the lake

Y = swimmer's income

T = travel costs incurred in reaching the lake

3. Socioeconomic Benefits

Information which can be used to specify the value to individual users (direct effects) and to the region (indirect effects) arising from various use levels and thus, ultimately to the water quality of the lake with and without some form of treatment.

4. Costs

Information which can be used to predict the cost of lake treatment and any necessary use facilities. Lake treatment costs are a function of the intensity of treatment and the treatment technology. Costs of facility development and maintenance will depend on the types of use anticipated as well as the number of users. The costs are thus dependent on many of the same variables as the water quality information.

FUNCTIONS VS. PROCEDURES

In all of the above, it would be nice to have functional relationships. To the extent this is possible the decision maker can insert the appropriate values of the independent variables into the functional equations, carry out the necessary arithmetic and obtain a valid measure of net benefit. However, in view of what was said above about benefit-cost analysis, and in recognition of the fact that the socioeconomic, and probably also the limnological relationships, vary so much from one part of the country to another, it is probably impossible to obtain generalized functions which can be used as described above. What is possible, is to develop a set of procedures (or models) which can be given empirical content for a given lake at a given time. The procedures need to be adaptable to situations where more or less money is available to finance them (presumably the precision will vary also). Furthermore, the procedures need to be elaborated, and enough general guidance given, so that the decision-making agency, perhaps with the aid of some consultation, can reasonably be expected to successfully use the procedures. It is the purpose of the research in the Clean Lakes Program to develop such procedures. Conceptually, the procedures can be put into four categories:

1. Procedures to identify as many of the social impacts as may exist in any given area.
2. Procedures to screen the impacts identified in order to determine those of sufficient importance to be analyzed in some depth.
3. Procedures to use in ascertaining the human consequences of the important impacts.
4. Procedures for determining the trade-off values of the important impacts.

PREDETERMINED CONDITIONS

Obtaining the information specified above through a research effort by EPA is conditioned by several items which appear to be in the nature of givens:

1. The budget available to the effort for the next year is limited to the $375,000 available for extramural funds, some travel and computer money for EPA employees, and the existing grants to Oregon State and the University of Wisconsin.
2. Part of the sociological research being conducted on the program is to be located at Liberty Lake near Spokane, Washington under a grant with Oregon State University. The remainder of the sociological research (as well as some economic research) is being conducted at White Clay, Mirror and Shadow Lakes in Wisconsin by the University of Wisconsin. It is desirable to integrate the sociological and economic aspects of the research.
3. Any significant amount of money for conducting in-house research is unavailable.
4. It is desirable to determine the degree to which user's perceptions of a lake's quality are correlated with limnological measures. This means that ideally any lake studied in the socioeconomic research would also have undergone a limnological evaluation. An acceptable substitute would be the availability of limnological analysis from another source if:
    a. the limnological analysis is consistent with those carried out in lakes undergoing limnological evaluation;
    b. the limnological data are sufficiently detailed to be useful in developing user demand and cost functions where they are logically dependent upon water quality changes.

4. The major thrust of the Clean Lakes Program demonstration projects to date is in restoring lakes for recreational (both physical and aesthetic) activity of one form or another. Therefore, for the research the social analysis should be confined to recreational use of lakes, deferring analysis of domestic and industrial consumption, irrigation, and other uses, unless there is some unavoidable connection with recreational use. However, in the longer run, if more money becomes available for social research, such other water uses should be considered for possible study.

SCOPE OF STUDY

In general, any given lake is part of a system of lakes which are substitutable, one for another, as bases for recreational activity. Furthermore, the several forms of recreational activity are not all compatible with one another in a confined area (e.g. swimming, waterskiing, sailing and hunting). Thus, the question of whether or not to restore a lake is really a whole series of questions including:

1. Which recreational uses are to be allocated to which lakes?
2. Which lakes are to be treated in one fashion or another?
3. What technique shall be used in treating each lake?
4. How intensively shall each lake be treated?

An important point about the questions above is that the answer to any one of them depends, to a considerable degree, upon the answers to all the others. Thus, ideally, the questions should be considered simultaneously in order to arrive at "good" answers. To do so requires that the four types of information needs described above (limnological and social) be considered for all the lakes in a region. Ideally, the boundaries of the region would be set such that it is a logical unit. Unfortunately, what is a logical unit in one sense may not be in another. From a limnological point of view, a logical unit may be a watershed. From an administrative point of view, a logical unit may be a governmental subdivision such as a village or county. And finally, from a use point of view, a logical unit may be a cluster of lakes lying in different governmental units and watersheds. The final definition of the region will need to be a compromise of the various viewpoints.

At this stage in the social research of the Clean Lakes Program it is clear that the extremes of complexity ought to be avoided. For example, the lakes around Seattle could be studied. But such a choice would be unwise because of the size and complexity of the region's economy and water based recreation activity. The latter is made all the more difficult by the presence of Puget Sound. A better approach would be to use less complicated regions to develop the procedures, then at a later date test their applicability to a region such as that around Seattle. On the other hand, a lake which is isolated from human concern has no social relationships and is thus not suitable for study. Between these extremes, there is a need to define the regional characteristics which should either be included in the research study regions or accounted for in some other way. One such characteristic is the availability of secondary data. The more such data are available the further the available research dollars can be stretched.

RESEARCH OBJECTIVES

There are several objectives which should be incorporated into the socioeconomic evaluation of the Clean Lakes Program. What follows is an enumeration of those objectives with some discussion of how each objective might be met.

1. Cost Information

The costs of changing the water quality parameters of a lake are of two sorts. The direct costs are those incurred in purchasing supplies, hiring labor, and using equipment to carry out a treatment such as dredging, spreading alum, or installing sewers along the shoreline. The indirect, or opportunity, costs are the decrease (if any) in benefits which occur as a result of the lake treatment. Consider a lake used by waterfowl for habitat. A dredging operation which reduces nutrients, and thus macrophytes and algae, may increase the benefits to swimmers. However, the loss of waterfowl may mean a decrease in benefits to hunters or bird watchers. Or, if land use regulation is judged to be an appropriate means to achieving lower nutrient inputs into a lake this may increase the costs, or decrease the income, of those who operate on the land. And, finally, there may be non-market costs which arise from the production of materials to be used in the lake manipulation. These costs might take the form of increased air pollution or energy use. Such costs could arise far from the region where the lake manipulation is being conducted.

For purposes of this statement costs will be used to mean direct costs as defined above. Indirect, or opportunity, costs will be discussed below under the heading of benefits.

Lake treatment cost data should be available as part of all the demonstration projects. This data can be used to obtain the required cost functions. However, caution is necessary: The costs incurred in the demonstration projects may run high compared to those of a routine procedure. Costs for facilities development should be readily available through standard engineering, architecture, and landscape design and estimating procedures.

2. User Demand Information

The reaction of users to any change in a lake's characteristics can be obtained in two ways. One is to ask them about their preferences through interviews, using questionnaires, photographs, etc. The second way is to observe their actions when faced with real life choices between lakes of different characteristics. Since there is evidence to the effect that people, particularily consumers, act differently in real life than they indicated they would in answering questions about preferences, actual observation is the better method where it is available.

Such observation will be available (at least to a degree) for users of lakes studied as part of this research program. However, it is not clear whether the results obtained from the study lakes will be transferrable to other lakes. If the underlying social influences are sufficiently diverse, the results will not be transferrable. If such is the case, then, observation is not available as a general procedure since all decisions about restoration must be made before their effects can be observed.

To determine the transferrability of the results of the lake restoration, any predictive model developed from observation of users' reactions at one lake should be tested by application to a second lake for which observations are also available.

Since the results obtained through observation may not be transferrable, it is desirable to use the current research to develop techniques for predicting user demand responses based on interviews, etc., before the restoration, and then to test and refine them by comparison with the results observed after restoration.

Because of the number of factors affecting recreationist's choices, considerable care must be used in establishing a research design that isolates changes in recreationist's actions due to differences in lake characteristics from changes due to differences in other factors. Thus, if observations of recreation use are taken before and after some lake treatment, care must be used to separate out the effects of possible differences in income, preferences or other circumstances of the recreation population. The same care must be exercised if the recreational uses of two different lakes are compared.

Research on user demand should begin by developing a list of the various user groups to be considered (swimmers, fishermen, etc.). Next a model, or perhaps several models, which are appropriate to the Clean Lakes Program can be identified. Such models should then be tested empirically through application to the several lakes chosen for study. The applications will require statistical analysis of site and user characteristics. To be useful in estimating the value of the lake to its users, the use demand information should contain an economic demand function. The social characteristics of the users (age, etc.) which are used as independent variables in such functions need to be identified. Such identification can be greatly assisted by close coordination between the economic and sociologic aspects of the research. Each test application will differ in locality and perhaps also in the lake treatment activities being used. And the investigator may be different. Consequently, there will need to be close coordination of the several grantees' work to insure the broadest applicability of the results.

3. Social Benefits

The social benefits arising from the presence of a lake can be grouped into five categories:

a. Those benefits accruing to individuals who use the lake for on-site activities such as swimming, hiking, viewing or residential living;

b. Those benefits accruing to individuals who may never visit the lake, but who value its existence either for the option of visiting it at some future time, or, perhaps, just because they consider the lake to be part of a good environment;

c. Those benefits accruing to people in the watershed whose land use activities are necessarily modified as part of the lake manipulation effort;

d. Secondary benefits arising in the region as a result of the patterns of attitudes, activities and economic trends. Such secondary benefits are derived from the primary benefits listed in items a - c above. The expenditures by on-site recreationists for travel, equipment, etc. are income for the suppliers. They, in turn, increase their expenditures and so on. In a similar way, the individuals whose attitudes and ideas are directly impacted by visits to the lake, or by the happy knowledge that it exists, affect the ideas, attitudes and activities of other individuals, creating additional impacts;

e. Any social benefits arising outside the region as the result of activities associated with lake restoration.

Each of the procedures identified in the preceding section on "Social Benefits" is described below:

a. Benefits to on-site users. One measure of benefits to on-site users is consumer's surplus. If the information on user demand is in functional form, with price as one of the independent variables, consumer surplus can be derived using standard procedures. This has been done in a number of recreation studies using such items as travel and on-site cost as proxies for price. In the case at hand, the recreational resource is frequently located close to potential users and provided essentially free of cost. Therefore, some proxy other than travel will need to be identified. Since any lake manipulation may work to the benefit of some users and to the detriment of others, there may well be both positive and negative elements of benefit.

  These benefits will accrue annually (though not necessarily at a constant rate). To compare these benefits with various other time streams of benefits and costs they should all be discounted to a common point in time. Therefore, it will be necessary to identify the appropriate rate of discount. That rate of discount may be determined either in legislative pronouncements by government or in the opportunity costs associated with using public capital.

  It is quite possible that the econometric analysis of on-site user demand will contain errors either in specifying the functional form of the equations or in specifying the independent variables for inclusion in the function. Although such errors can be reduced by careful analysis and coordination between the economic and sociological aspects of the study it is important to recognize that such errors should be identified and interpreted as to their consequences. Here again, close cooperation can be helpful.

b. Procedures to identify the benefits to individuals who do not actually visit the lake, but who value it either for the option of future use, or as simply part of a good environment, need to be identified. Such benefits may be sizeable or insignificant in any given case. To the extent that such benefits are significant, there is a need to not only identify them in qualitative terms but also to develop procedures analagous to consumer surplus (as described above), which can be used to quantify as well as possible the values involved.

c. Costs of land use modification. If land use modifications in the drainage basin are part of the lake manipulation project, they may lead to either increased costs or decreased incomes to the individuals who must change their activities. Such results can be of many different sorts. Examples are installing sewers to avoid ground water contimination, and changing agricultural and forestry techniques to reduce siltation. At this time it is impossible to anticipate all such possibilities. What is needed is a systematic process to identify whether or not such impacts exist and, if so, how to analyze them.

d. Regional benefits. In the short run, changed levels of economic activity in the region will arise only if exports of regional products (including recreation) are increased, or if regional imports are decreased. In most cases, lake modification will not lead to creation of any significant increase in export of regional product. Such an increase would only arise by attracting recreationists from outside the region. Unless the recreational opportunities to be enhanced are sufficiently unique to attract outsiders, in spite of the travel and other costs involved the increase in exports will be negligible.

Decreasing regional imports of recreation may occur if the residents of the region substitute local recreation sites for sites outside the region. This may probably occur to some extent, but may or may not be very significant. What is likely to occur is a shift in leisure time activities and attitudes of the regional residents. Whether this will lead to observable changes in the level of economic activity is uncertain.

In the longer run, the regional impacts of environmental improvement including lake improvement is to make the region a better place to live. This may lead to increases in population and the location of new firms (or it may avoid decreases). How significant this effect is likely to be will depend on a number of other characteristics of the region.

The models for ascertaining any regional impacts in economic activity are economic base and input-output models. They are well developed conceptually and need only be structured to fit the needs of the Clean Lakes Program. Their use in the research should be to answer two questions: 1) what are the circumstances, if any, in which regional impacts of lake manipulation are significant, and 2) what is the most efficient way to structure the models to identify regional impacts at various budget levels for data collection.

The significant regional social impacts which are not contained in the economic impact as described above need to be enumerated, described, and quantified where feasible. One obvious question concerns who it is that received the benefits, and who bears the costs (both financial and non-financial). Some of this information may be obtained in terms of income and employment. However, there is need to identify any possible other secondary social impacts, such as an increase (or decrease) in community cooperation or a shift in what Smith and Hogg[3] term the benefactors and beneficiaries.

e. Impacts beyond the region. The alternative lake restoration techniques used need to be reviewed to ascertain whether they require equipment or materials which have significant impacts that are not incorporated in the market prices involved in their production. If they do, the nature of those impacts should be identified and measured as well as possible.

4. Allocation Procedures

If a region contains several lakes and the best allocation of recreational uses and resources is not obvious, then some means to identify or at least approximate, the best allocation is needed. Any particular allocation implies certain use levels for each lake with the consequent benefits and costs. Transportation costs of the consumer may well vary from one allocation to another. Public expenditure for transportation facilities, recreational development and police will possibly differ. To the extent that two allocations imply different patterns of residential development, costs for the whole gamut of social services may also differ.

In such a situation, an allocation model is needed. The model can be used to both identify alternative allocations which should be given serious consideration as well as to estimate the social and private resources which will be needed to implement the alternatives. There are several such allocation models available. Those best suited need to be identified and structured to meet the needs of the Clean Lakes Program.

PHASES OF WORK

The work should be carried out in three phases: (1) Establishing a conceptual framework; (2) initial application of the framework to specific cases using such empirical data as can be readily obtained, and (3) refining the relationships obtained in phase (2) through replication of study areas and also through longer term study.

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[3] Smith, Courtland L. and Thomas C. Hogg. 1971. Benefits and Beneficiaries: Contrasting Economic and Cultural Distinctions. Water Resources Research 7(2):254-263.

The conceptual framework will provide a guide for the specific analysis in phases (2) and (3). Phase (2) will provide a quick (and hopefully not too dirty) estimate of the results to be obtained in any actual case of lake manipulation. Thus, decision makers will have a basis for making relatively informed decisions regarding lake manipulation. Phase (3) will provide a more precise basis for such decisions. As in all research work, there will be feedbacks from phases (2) and (3) about the appropriate conceptual framework. Phase (2) will be particularly important in this regard if the conceptual framework is to be sufficiently detailed and specific to be useful in real life decision making.

Phase (1) will require:

1. Literature searches.
2. Discussion with knowledgeable individuals involved in the Clean Lakes Program.
3. Selection of research regions.
4. The development of a theoretical formulation of the various models to be used.

Target date for completion: June 1978

Phase (2) will require:

1. Collection of primary and secondary data including sampling considerations.
2. Analysis of the data using the theoretical formulation developed in Phase (1).
3. Review and revision of the theoretical formulation

Target date for completion: June 1979

Phase (3) will essentially duplicate Phase (2) but in a more thorough manner. Phase (3) should be partially completed upon termination of the grants to be funded with current monies, i.e., in 1981. However, the research envisioned with the current funds will be insufficient to carry phase (3) very far.

THE CHANGING POLITICS OF WATER POLLUTION CONTROL

by

G. J. Protasel*

THE EMERGENCE OF REGULATORY POLITICS

Two significant political transformations have gradually evolved in the United States which have had a marked effect on water pollution control efforts. First of all, new political demands for water pollution control have come on the scene. One has witnessed the rise and proliferation of environmental groups, not just at the national, but also at the local level. Secondly, centralized governmental control has become an important factor in water pollution control. Historically, there has been a pronounced shift from local to national controls.

These two political transformations have come about largely because of the increasing interdependency of American society. The scope and nature of the water pollution problem has necessitated control by larger jurisdictions. Local communities that traditionally had responsibility for clean water found themselves unable to exercise their responsibility effectively. Downstream residents could not control the behavior of upstream residents. As a result, water pollution control has become centralized.[1]

As American society has become more complex and interdependent, there has also been an increased awareness of the negative side effects of economic growth. Private decisions have been sometimes seen to produce "collective bads" such as water pollution. The fact that the natural ecosystem was often ruthlessly exploited led to the generation of new political demands to halt environmental degradation.

These transformations (environmentalist demands and centralized pollution control), that have resulted from the increased interdependencies of the American economy and society, have changed the politics of water pollution control. First of all, government now exercises more coercion than in the past. The national government now exercises more immediate control over the conduct of individual and corporate behavior than ever before in the area of water pollution control. Secondly, the decision-making and policy-making process has become more conflictual. Environmental interests are no longer willing to tolerate the laissez-faire doctrine of non-interference in private decisions, which have produced large social costs in the past. This cleavage between development and environmental protection interests is well known.

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Increased governmental coercion and increased decision-making conflict indicate that water pollution control policy has entered the "regulatory arena" of politics.[2] The pattern of politics found in the regulatory area fits the pluralist pressure-group description of American politics. Coalitions of common interest are forged only after much bargaining and logrolling have taken place amongst interest groups. Environmentalist and developmentalist interests are eventually accommodated, but only after some decision-making conflict.

In the regulatory arena the government acts essentially as an umpire for the bargaining organized interests (unorganized interests are kept out of the decision-making process). Regulations governing individual and group behavior are the outcome of the pluralistic bargaining process. This interest group bargaining is facilitated by broad legislative mandates which permit the details of regulation to be worked out to fit the needs of the interests affected. A combination of symbolic politics and organizational capture thus characterizes regulatory politics.

Government regulation and decision-making conflict are basic facts of life in the area of water pollution control policy. One should recognize, however, that this particular pattern of politics shows signs of instability. Unless one fully comprehends the sources of the instability, one is likely to ignore important undercurrents of politics that might significantly alter the politics of water pollution control. The instability of regulatory politics is discussed below.

THE INSTABILITIES OF REGULATORY POLITICS

In one sense, regulatory politics is inherently unstable because it relies on a bargaining process to form a common interest. The common interest is subject to continual redefinition as support for different winning coalitions waxes and wanes in the process of bargaining. There are other sources of instability, however, which though exogeneous to the bargaining process threaten to disrupt the regulatory pattern of politics. These sources of instability are outlined below.

Reluctance of environmentalist groups to bargain.

There are times when environmentalist groups will take an intransigent Nader-like stance and refuse to bargain.[3] Needless to say, failure to adhere to the regulatory pattern of politics (bargaining and logrolling) expands the level of decision-making conflict. Emphasis on the physical parameters of the ecosystem shifts the political debate from the individual to the systems perspective. The policy-making arena becomes filled with ideological pleas to readjust life styles to the era of the spaceship earth. The multiplicity of sides which the pollution issue had in the regulatory arena is reduced to just two sides--development/protection. The need for governmental coercion is stressed. Certainly leviathan is preferable to oblivion.

This ideological expansion of policy-making conflict from the group to the systems level, with the expectation that governmental coercion is necessary to resolve the conflict, signals a shift from the regulatory to the redistributive arena of politics. The message is that governmental policy must be designed to cause society to fundamentally alter its behavior, or else face ecological disaster.

Reluctance of developmentalists to alter behavior.

While environmentalists sometimes refuse to participate in the regulatory bargaining process, developmentalists sometimes refuse to recognize environmentalists' interests. Some developmentalists certainly would prefer to shun the regulatory arena altogether and instead embrace the old doctrine of mutual-noninterference, which previously dominated water resources politics. Under the doctrine of mutual-noninterference, coalitions of common interests would not have to be formed with environmentalist or other interests. Instead coalitions of uncommon interests would prevail which would eliminate the need for interest group bargaining. Individual and corporate interests would simply try to get what each could from government in the way of favorable policy treatment. Pork barrel legislation such as that authorizing the construction of dams by the Army Corps of Engineers exemplifies this old type of distributive politics.

Of course, it is extremely unlikely that the U.S. could ever go back completely to the old distributive politics where environmental concerns were sacrificed under the doctrine of mutual-noninterference. Modern technology has made it possible though for a new type of distributive politics, a new privatization of the public interest, to occur. It is technologically feasible to centrally collect and process polluted water. Environmental Protection Agency grants to construct water treatment facilities, for example, permit the maintenance of a coalition of uncommon interests so characteristic of distributive politics.[4] Technology achieves the water quality standards the environmentalists want without requiring the polluters to change their private behavior. Both public and private interests are seemingly protected by this new distributive politics.

As with old distributive politics, the new distributive politics occurs without regard for costs in the short run. In the long run, one wonders if there will be enough money for construction projects to meet the growing demands for pollution control. Will technological break-throughs occur which will make the new distributive politics even more cost-effective?

Even if one believes that technology will be able to solve the pollution problem, the pressing question remains as to whether the public sector should have to shoulder alone the burden and cost of pollution cleanup? Perhaps a strategy which would provide polluters with incentives to change behavior would be more effective public policy. One needs only to compare the efficacy of water and air pollution control expenditures to see the advantages of such a strategy.[5] Unlike polluted water which can be centrally collected and treated, polluted air does not lend itself to a centralized treatment approach. Air pollution control efforts have thus been forced by the nature of the pollution problem to be directed at the sources of pollution. In comparison with construction projects designed to treat polluted water, air pollution control efforts have made more progress.

The old adage that an ounce of prevention is worth a pound of cure is a message that might make the new distributive politics less stable than the old distributive politics. As budgetary pressures mount and uncertainties about technological breakthroughs arise, there may be pressures to abandon the new distributive pattern of politics.

Pressures to eliminate governmental red tape and make government more responsive to the public.

Regulatory politics is accompanied by a proliferation of rules and procedures. This regulatory red tape is a direct result of the broad mandates which Congress gives administrative agencies. Regulatory legislation is purposely broad and symbolic in nature to accomodate administrative rulemaking and secure the support of the public. The effectiveness and legitimacy of the government is impaired, however, whenever the red tape becomes excessive and the public becomes disenchanted with the government's ability to deliver on its promises.

Symbolic politics and the proliferation of regulation are results of the extreme difficulty of directly controlling individual behavior in a vast complex society. The fact that individual actors find multitudinous ways of evading the "stick" approach to government further exacerbates the problem. Consequently, there has been considerable discussion that governmental policy should be formulated more along the lines of the "carrot" approach i.e. manipulate the rules of conduct which govern behavior modification, rather than controlling it directly.[6] Social contact through behavior modification, rather than through direct governmental coercion, seems to have certain advantages over the regulatory approach. The deliberate manipulation of the environment of conduct by government characterizes what will be termed the self-regulatory arena of politics.

Self-regulatory politics is distinguished by efforts to design appropriate incentives, penalties, and rewards which minimize the need for governmental coercion. Government control is exerted indirectly through a reliance on user charges, effluent taxes, etc., rather than directly through regulations. It should be noted that the design of appropriate incentives probably minimizes the phenomenon of organizational capture that occurs so often in the regulatory arena. Pressure group politics no longer dominates the governmental agency. Government no longer looks on as a spectator or umpire, but instead takes the role of an active manipulator. Self-regulation is a structural rather than a procedural approach to policy making and as a result the strength of interest groups is diluted.

While the self-regulation strategy undoubtedly reduces the red tape that is associated with administrative-rule making, the manipulation of incentive systems accentuates the need for governmental responsiveness. Manipulation of incentives must be done in the name of the public interest. The self-regulation strategy thus makes the problem of defining the public interest more critical than ever before. The design of appropriate incentives requires consensus regarding the legitimate scope and direction of governmental activities.

The other side of self-regulatory policy-making concerns the manipulation of the symbolic environment in order to build the social consensus that is necessary to legitimize governmental intervention. Manipulation of the symbolic environment in the self-regulatory arena is different, however, from the symbolic politics which commonly occurs in the regulatory arena. Here we have systematic effort at government propaganda. This is not merely the erection of a symbolic screen behind which powerful organized interests bargain, but represents a geniune effort to actively and positively direct the efforts of government. The social indicators movement typifies these efforts.

At a time when it is said that the nation is suffering from an identity crisis, and when the multiplication of subgroups appears to threaten to carve up the body politic, there are pressures to achieve governmental unity by focusing the public's attention on common points of reference.[7] The idea of a social performance index for corporations which is strongly resisted by industry seems to be a prerequisite for responsive self-regulatory politics. Less government is feasible only if there is some way to monitor and shape corporate performance in the public interest. If society wants to avoid the high costs of regulation, it seems that industry might be forced to bear the burden of performance evaluation by government.

EXPLAINING CHANGES IN PATTERNS OF POLITICS: A THEORETICAL CONSTRUCT

The politics of water pollution control have been shown to be dominated by the emergence of regulatory politics. The instabilities of the regulatory pattern of politics have been described above. Illustrations have been made of how the regulatory pattern of politics might easily give way to three other patterns of politics--redistributive, distributive, or self-regulatory. In this section factors which cause the patterns of politics to shift are more systematically examined.

Before one can examine the forces which may produce different patterns of politics one needs to agree on a schema for identifying the public policy arenas which invariably generate the different patterns of politics. The following typology based primarily on Theodore Lowi's framework seems useful for this purpose.[8]

		Likelihood of Coercion	
		Remote coercion	Immediate coercion
Applicability of Coercion	Individual conduct	DISTRIBUTIVE	REGULATORY
	Environment of conduct	SELF-REGULATORY	REDISTRIBUTIVE

Figure 1. Typology of public policies.

The public policies are classified according to two dimensions--likelihood of coercion and applicability of coercion. Distributive and regulatory policies are similar to the extent that government coercion acts directly on the individual or group, but are different to the degree that distributive politics are less coercive than regulatory policies. Likewise, redistributive policies are more immediately coercive than self-regulatory policies, but are similar in that coercion acts through the environment of conduct. Of course, it should be remembered that in the final analysis all government is somewhat coercive.

It should be noted that each of the policies is associated with a particular pattern of politics (described in previous sections). Policies determine politics. Policy is the independent variable and politics the dependent variable. The prevailing policy definition shapes the decision-making agenda by shaping political actions. Political behavior is thus structured by policy expectations. Below two propositions are set forth which attempt to explain why policy arenas, and thereby patterns of politics, change.

Proposition One.

Increases in social interdependencies increase the likelihood of governmental coercion.

As society becomes more interdependent the external costs of private actions increase which bring about pressures for governmental coercion. Whenever private decisions begin to play havoc with the public interest, the initial governmental response to any such public bad is to handle it with a dose of immediate coercion. The image that many people have of government as the ultimate authority perhaps explains why there is so much governmental regulation. Nothing seems more just than direct governmental regulation of individual or group behavior that threatens the public interest.

Proposition Two.

Increases in decision-making costs increase the likelihood that governmental coercion will be applied through the environment of conduct rather than individual conduct.

As the costs of governmental decision-making increase, there is an incentive to cut costs by indirectly applying governmental coercion through the environment rather than through individuals and groups directly.[9] Disenchantment with red tape and the cost of governemnt regulation spawn searches for new types of governmental intervention which don't get bogged down in trying to control individual behavior. Increased costs of information and control encourage more indirect methods of social control.

Graphically the determinants of patterns of politics can be displayed as follows:

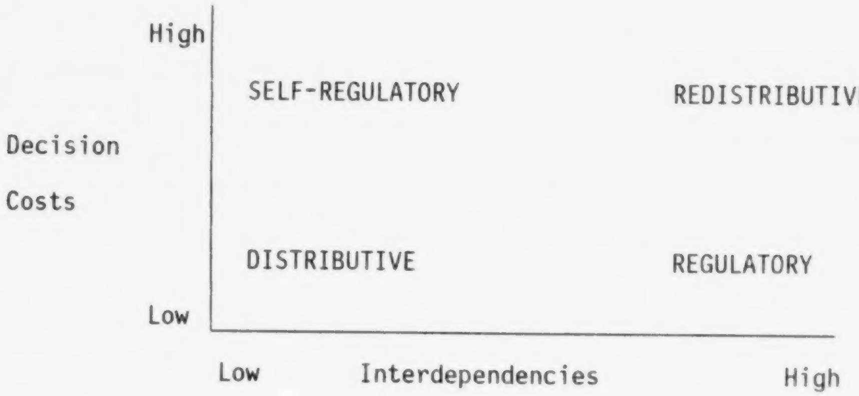

Figure 2. Theory of policy change.

POLICY PERSPECTIVES AND THE EPA CLEAN LAKE PROGRAM

Up to this point this paper has been concerned with the impact that patterns of politics have on the water pollution control agenda and with the forces that shape policy arenas, which in turn structure patterns of politics. In the previous section, attention was given to factors which may shape policy arenas in the long run - social interdependencies and decision-making costs. In this section, attention now shifts to factors which may also shape policy arenas, but in the short run. The focus will be on the policy perspectives of decision-makers at different levels in the intergovernmental system and how their perceptions are likely to affect the implementation of the Clean Lake Program.

No single pattern of politics may necessarily dominate the intergovernmental decision-making arena. Different actors at different points in the federal system are concerned with different facets of politics and perceive the water pollution control agenda in different ways. For ease of discussion, the analysis will be limited to the contrast between national and local political perspectives.

It is believed that national decision-makers tend to look at the problem of lake eutrophication from a structural perspective. Their concern is with finding a structural or institutional solution to the deteriorating lake problem. Their efforts center on eradicating a public bad, not providing a public good. In terms of the previous discussion, this implies that national policy makers tend to view the task of cleaning up the lake in terms of regulatory or self-regulatory policy.

In contrast, local officials are likely to have an allocative perspective of the Clean Lake Program. Local decision-makers are concerned with the allocative consequences of cleaning up the lake, i.e., who benefits and who loses. For this reason the deteriorating lake problem is likely to be defined in terms of distributive or redistributive policy. As one will see below, the fact that the issue of lake cleanup is hard to separate from the issue of land use tends to accentuate this allocative dimension.

The Structural Perspective

The EPA Clean Lake Program (Section 314 of the 1972 Water Pollution Control Act Amendments) provides grants for demonstration projects designed to combat non-point source pollution. This focus on demonstration projects stems from a recognition of the fact that the nature of non-point source pollution makes the traditional EPA regulatory strategy difficult to carry out. Every lake is a unique ecosystem. Nonpoint source pollution is caused by many different factors not subject to uniform technological control. Controlling nonpoint source pollution thus demands flexible policy making.

The regulatory approach that is successful for halting pollution as it comes out of the pipe may be too burdensome to control non-point source pollution. After all, a regulatory strategy implies the ability to detect and punish individual violators. Non-point source pollution makes this immediate exercise of coercion through the individual rather difficult. Usually regulatory politics is dominated by a few large organized interests who are responsible for most of the pollution. In the case of non-point source pollution, however, the pollution stems from the entire community. Regulating a community is a different task from regulating a polluting industry.

In short, there are reasons to believe that a traditional regulatory approach will not work successfully in cleaning up non-point source pollution of lakes. Many of the ingredients for effective regulatory policy are missing, such as clearcut standards, proven technology, and readily identifiable sources of pollution.

From a structural perspective there appear to be mounting pressures which discourage a strict regulatory approach to non-point source water pollution control. A self-regulatory or community development strategy thus becomes as an attractive alternative. The chief problem with a self-regulatory strategy is that it relies upon an appropriate unit of government to take the responsibility for clean lake management. Oftentimes existing governmental jurisdictions (state, city, county or special district) may not be particularly interested in or capable of lake management. Effective lake management may thus require the design of new units of government capable of dealing directly with the problem of eutrophication.[10]

The theory of collective good may be used to provide guidelines for decision-makers with a structural perspective who wish to design lake management districts. According to the theory, for a collective good to be provided at an optimal level there must be a match between the boundaries of the collective good and the boundaries of the governmental jurisdiction which provides the good or service.[11]

If benefits spill over governmental boundaries then there will be disincentive to produce the good or service at an optimal level. For example, if a cleaner lake attracts greater numbers of the public from outside of the lakeside governmental jurisdiction (tourists, fishermen, swimmers, skiers, etc.) then this may act as a disincentive for lake cleanup. Under such circumstances, less than an optimal level of water pollution control may be reached. Grants-in-aid are frequently used to remedy such a situation. The intergovernmental transfer payments compensate the unit of government for producing benefits which spill over jurisdictional boundaries. The EPA Lake Restoration Grant Program illustrates such an approach.

The problem of benefit spillovers is not the only type of mismatch between the boundaries of the collective good and the boundaries of the governmental jurisdiction which can occur. When the boundaries of the governmental jurisdiction are larger than the boundaries of the collective good there is also a disincentive to produce an optimal amount of the good. For example, if the primary beneficiaries of a cleaner lake are the landowners immediately surrounding the lake then this may act as a disincentive for a governmental jurisdiction, which is large enough to contain many other elements of the public, to clean up the lake. The majority of citizens may not want to pay the full cost of lake restoration which would benefit a minority of lakeside residents. The governmental jurisdictions which overlap the lake's boundaries may simply be too large to provide an optimal level of the collective good.

More often than not, it would seem that the problem of lake restoration centers around the problem of how to deal with overlapping governmental jurisdictions which are too large, rather than the problem of benefit spillovers, which result from governmental jurisdictions which are too small. If this is the case, then certain questions need to be raised about the effectiveness of a grant-in-aid approach to encourage lake restoration. There would seem to be an asymmetry of grant-in-aid effectiveness that would depend on whether the less than optimal provision of a collective good was the result of governmental jurisdictions that were "too small" or "too large". Grants-in-aid are most effective in dealing with benefits spillovers which flow from governmental jurisdictions that are smaller than the boundaries of the collective good.

In this situation, the level of provision of the collective good can be increased with a minimum of organizational effort. An existing organization simply expands its production. On the other hand, when the unit of government is larger than the boundaries of the collective good, a larger amount of organizational inertia will have to be overcome before the amount of the collective good produced will increase; perhaps requiring a larger grant-in-aid stimulus. Oftentimes, a new subunit will have to be created to deal with the problem, which further complicates the undertaking.

If self-regulation is to work, there must be appropriate size jurisdictions for lake management. To achieve a close match between the boundaries of the collective good and the boundaries of the governmental jurisdiction probably will require more government rather than less. A community development strategy whereby the potential beneficiaries of lake restoration are given the opportunity to organize and design a lake management district, whose boundaries closely correspond with the boundaries of the collective good, seems to be a viable policy option. It appears that the institutional aspects of lake management must first be ironed out before the traditional grant-in-aid approach will act as an effective incentive for lake restoration.

The Allocative Perspective

At the local level, decision-makers are especially concerned with the allocative consequences of cleaning up the lake. The question of who benefits and who loses as a result of lake restoration is likely to be paramount in the public's mind as well. There are some factors which suggest that controlling non-point source lake pollution would be perceived as a distributive outcome i.e., each individual benefiting, while costs are shared by the entire community.

First of all, the costs of non-point pollution control are relatively diffuse. In contrast to point source pollution control, where the costs of cleanup would be paid by a few large industrial polluters, the costs of non-point source pollution control are more evenly spread throughout the community. It is therefore less likely that one will have a situation where a large organized interest will object to the costs of pollution control and try to keep it off the political agenda.[12]

Secondly, it appears that lake restoration provides benefits to a wide range of interests. Indeed, it could be argued that both development and protectionist interests benefit from a clean lake. After all, a clean lake enhances property values of the surrounding area, facilitating further development of the region.

It is the difficulty of separating the issue of clean lakes from the issue of land use control, however, that is likely to move the water pollution control efforts into a redistributive policy context i.e., a direct conflict between environmentalists and developmentalists. It has often been the case that no-growth minded citizens have taken up the environmentalist banner to halt development. In the case of lake restoration efforts this familiar strategy will not necessarily work to the advantage of those who wish to curtail growth. For example, cleaning up the lake, say by installing a sewer system, is likely to make the lakeside area a more attractive place for development.

Those who fear development might be expected to point to the "crowding effect" that a cleaner lake might generate. The last thing that a no-growth advocate wants is for the lake to be turned into a tourist haven, or become a population center. No-growth advocates and other protectors of community life-styles may thus find themselves in the strange situation of being on the opposing side of an environmental issue. On the other hand, growth proponents may be able to use the environmental issue to their advantage. Support for water pollution control is not necessarily synonymous with a no-growth philosophy.

In conclusion, it can be said that support for land use control may really underlie the politics of lake restoration at the local level. Prevailing community attitudes toward growth might very well determine where on the continuum of distributive-redistributive politics the issue of lake restoration will lie. Lake restoration efforts will be successful to the extent that the issue of the clean lake can be separated from the development issue. A lake management district without any police powers of land use control would seem to be the type of institution that would be needed to withstand the conflicts of redistributive politics.

NOTES

1. It should be pointed out that the centralization of water pollution control did not really cause local governments to lose power in the federal system. No real loss of local power occurred because local communities were incapable of exercising their responsibility to begin with.

2. For vivid analytical descriptions of regulatory politics see Theodore Lowi, "American Business, Public Policy, Case-Studies and Political Theory", World Politics, 16 (July 1964), pp 677-715; and Murray Edelman, The Symbolic Uses of Politics, Chapter 3, Urbana: University of Illinois Press, 1964, pp. 44-72.

3. For a discussion of the political art of compromise as it affects water resource policy making see Daniel M. Ogden, Jr., "The Real World of Political Decision-Making in Water Resources Policy", Treatise on Urban Water Systems, Maurice L. Albertson, L. Scott Tucker, and Donald C. Taylor, eds., Colorado State University, Fort Collins, July 1971, pp. 740-752.

4. Helen Ingram and J. R. McCain, "Federal Water Resources Management: The Administrative Setting", Public Administration Review, September/October 1977, Vol. 37, No. 5, pp. 448-455.

5. Air and water pollution are skillfully compared in J. Clarence Davies III and Barbara S. Davies, The Politics of Pollution, Indianapolis, Indiana: The Bobbs-Merrill Company, Inc., 1976, second edition, pp. 23-25.

6. Charles L. Schultze's argument that government should rely more on marketlike incentives to further public policy ends states this point of view well. See Charles L. Shultze, The Public Use of Private Interest, Washington, D.C.: The Brookings Institution, 1977.

7. The need to have a system of social indicators which would provide a common focus of attention for policy makers and the public is described in R. D. Brunner and J. P. Crecine, "The Impact of Communication Technology on Government: A Developmental Construct" paper prepared for delivery at the 1971 Annual Meeting of the American Political Science Association, Conrad Hilton Hotel, Chicago, Illinois, September 7-11.

8. This typology is based on Lowi's except that it inserts Salisbury and Heinz's notion of self-regulatory policy in place of Lowi's constituent policy. See Theodore J. Lowi, "Four Systems of Policy, Politics and Choice", Public Administration Review, July/Aug. 1972, pp. 298-310;

Robert Salisbury and John Heinz, "A Theory of Policy Analysis and Some Preliminary Applications" in Policy Analysis in Political Science, ed. by Ira Sharkansky, San Francisco: Markham, 1970, pp. 39-60.

9. It should be noted that the concern here is for the costs of decision-making within particular policy arenas. This differs from Salisbury and Heinz's treatment of decision costs. For example, according to Salisbury and Heinz's theory, high decision-making costs might result in the passage of regulatory policy. This clearly contrasts with the notion that high costs of decision-making cause movement away from the regulatory policy arena.

10. Lowell L. Klessig, "Open Marriage: Community Development and Environmental Management", Journal of Extension, Vol. XV, Sept./Oct. 1977, pp. 6-11.

11. Mancur Olson, Jr., "The Optimal Allocation of Jurisdictional Responsibility: The Principle of 'Fiscal Equivalence'", in The Analysis and Evaluation of Public Expenditures: The PPB System, U.S. Congress, Joint Economic Committee, 91st Congress, 1st. Session, 1969, pp. 321-331.

12. An analysis of how organized interests are capable of keeping pollution control off of the policy making agenda is contained in Matthew A. Crenson, The Un-Politics of Air Pollution, Baltimore: The John Hopkins University Press, 1971.